



Natural Resource Conditions at Mesa Verde National Park

Findings and Management Considerations for Selected Resources



View down Navajo Canyon.
NPS / KAYLA MCCRAREN

Natural resource conditions at Mesa Verde National Park: Findings and management considerations for selected resources

Science Report NPS/SR—2025/295

Allison Jones¹, Sarah McTague¹, Tina Mozelewski¹, Rachel Crafford², Andy Maguire¹, Justin Suraci¹

¹ Conservation Science Partners
Truckee, California

² Western Colorado University
Gunnison, Colorado

Please cite this publication as:

Jones, A., S. McTague, T. Mozelewski, R. Crafford, A. Maguire, and J. Suraci. 2025. Natural resource conditions at Mesa Verde National Park: Findings and management considerations for selected resources. Science Report NPS/SR—2025/295. National Park Service, Fort Collins, Colorado. <https://doi.org/10.36967/2311540>

The National Park Service Science Report Series disseminates information, analysis, and results of scientific studies and related topics concerning resources and lands managed by the National Park Service. The series supports the advancement of science, informed decisions, and the achievement of the National Park Service mission.

All manuscripts in the series receive the appropriate level of peer review to ensure that the information is scientifically credible and technically accurate.

Views, statements, findings, conclusions, recommendations, and data in this report do not necessarily reflect views and policies of the National Park Service, US Department of the Interior. Mention of trade names or commercial products does not constitute endorsement or recommendation for use by the US Government.

The Department of the Interior protects and manages the nation's natural resources and cultural heritage; provides scientific and other information about those resources; and honors its special responsibilities to American Indians, Alaska Natives, and affiliated Island Communities.

This report is available in digital format from the [National Park Service DataStore](#) and the [Natural Resource Publications Management website](#). If you have difficulty accessing information in this publication, particularly if using assistive technology, please email irma@nps.gov.

Contents

	Page
Figures.....	vi
Tables.....	x
Appendices.....	xii
Abstract.....	xiii
Executive Summary	xiv
Acknowledgments.....	xvii
Glossary	xix
Natural Resource Condition Assessment—Introduction	1
Conceptual Framework for Assessing Resource Conditions and Influences	1
NRCA Goals.....	2
Technical Report	2
Chapter 1. Introduction to Park Setting and Resources	3
1.1 Location.....	3
1.2 MEVE Purpose, Significance and Fundamental Resources and Values	5
1.3 Climate	6
1.4 Unique Ecological Qualities of Mesa Verde	6
Chapter 2. Drivers and Stressors.....	8
2.1 Climate Change	9
2.2 Increased Incidence of Fire	12
2.3 Spread of Invasive Non-native Plants	13
Chapter 3. Focal Resource Evaluations	15
3.1. Wildlife—Avifauna (Raptors, Passerines, and Hummingbirds)	15
Raptors Gap Analysis Summary	17
Passerines Condition Assessment Summary	23
Hummingbirds Condition Assessment Summary	30
3.2. Wildlife—Deer and Elk.....	32
Deer and Elk Gap Analysis Summary.....	32
3.3. Rare Plants.....	36

Contents (continued)

	Page
Rare Plants Gap Analysis Summary.....	37
Cliff Palace Milkvetch Gap Analysis Summary	43
Chapin Mesa Milkvetch Condition Assessment Summary	47
3.4. Mancos River (Water Quality, Macroinvertebrates, Fish, Riparian Corridor	54
Condition Assessment Summary for Water Quality	56
Condition Assessment Summary for Macroinvertebrates and their habitat	59
Condition Assessment Summary for Fishes and Aquatic Habitat.....	65
Condition Assessment Summary for Riparian and River Corridor.....	71
3.5. Upland Vegetation System	73
Upland Vegetation Gap Analysis Summaries	76
Piñon-Juniper (PJ) Woodlands Condition Assessment Summary.....	83
3.6 Seeps and springs	90
Seeps and Springs Gap Analysis Summary.....	91
Chapter 4. Management Considerations	95
4.1 Avifauna Monitoring and Management Recommendations.....	95
Passerines	95
Recommendation for Future Studies to Support Management	95
4.2 Deer and Elk Monitoring and Management Recommendations	96
4.3 Rare Plants Monitoring and Management Recommendations	97
Chapin Mesa Milkvetch	97
Cliff Palace Milkvetch.....	98
Recommendation for Future Studies and Monitoring (All Species) to Support Management	98
4.4 Mancos River Monitoring and Management Recommendations	99
Recommendation for Future Studies to Support Management	100
4.5 Upland Vegetation Monitoring and Management Recommendations	101
Piñon-juniper Woodland	101
Non-native Species.....	102

Contents (continued)

	Page
Fire, and Post-Fire Monitoring	103
4.6 Seeps and Springs Monitoring and Management Recommendations	104
Recommendation for Future Studies to Support Management	104
Literature Cited	105

Figures

	Page
Figure 1. The drivers, pressures, stressors, states, responses (DPSSR) logic framework used to organize NRCA report content.	1
Figure 2. NPS map of MEVE and the surrounding region.	4
Figure 3. a) Historical trends of yearly temperatures and b) yearly precipitation from 1895 to 2022 for MEVE.	10
Figure 4. Change in a) monthly average precipitation and b) monthly average temperature in 2050, relative to historical conditions, based on two climate scenarios:	11
Figure 5. Indicators of condition rating classes and colors.	15
Figure 6. Golden eagle (GE) detections per year from 1932 to 2001 based on MEVE records.	18
Figure 7. Predicted relative density of golden eagle nest sites, based on spatial layers developed by USFWS.	19
Figure 8. Mexican Spotted Owl (MSO) detections per year from 1980 to 2005 based on MEVE records.	22
Figure 9. Sites where Mexican spotted owls have been detected in MEVE, color coded for the decade of last detection..	23
Figure 10. Density per hectare of most common passerine species in MEVE.	24
Figure 11. Species richness (number of total species) per year from 2007 to 2022 based on data from the IMBCR and the SCPN.	26
Figure 12. Average occupancy rates (from 2008 to 2022) of 40 common bird species in MEVE (CO-BCR16-SC, shown in blue) compared to average occupancy rates for those species for the same time period in the broader Colorado-portion of the Southern Rockies/Colorado Plateau region (CO-BCR 16, shown in orange).	27
Figure 13. Effects of vegetation structure and composition and fire coefficients on breeding season abundances for 14 common MEVE passerines.	29
Figure 14. Annual count data of broad-tailed hummingbirds and black-chinned hummingbirds banded in MEVE from 2007 to 2022.	31
Figure 15. Annual estimation of deer and elk herd sizes (for the herd in the Game Management unit that includes MEVE) from 2004 to 2022 compared to annual precipitation (in).....	34
Figure 16. Overlay of elk and mule deer migration corridors and winter and summer ranges within CPW Data Analysis Unit (DAU) E24, which consists of Game Management Units 70, 71, 711, 72, and 73 and DAU D29.	35

Figures (continued)

	Page
Figure 17. Surveyed locations of 14 species of rare plants within MEVE.	39
Figure 18. Survey locations of 11 species of rare plants within MEVE overlaid with historical fires ≥ 10 acres from 1970 and onward.	40
Figure 19. Survey locations of 11 species of rare plants within MEVE overlaid with the percent cover of annual grasses and forbs in 2023.	42
Figure 20. Known range of Cliff Palace milkvetch (CPM) and Chapin Mesa milkvetch (CMM) within MEVE.....	44
Figure 21. a) Density of CPM (individuals / m ²) in 2003, 2004, 2005, 2007, 2021 and 2022 in burned and unburned transects.....	46
Figure 22. Density (individuals / m ²) of emerged CMM individuals of all age classes in 2001, 2003, and 2011 to 2014 in burned and unburned transects.....	48
Figure 23. Density (individuals / m ²) of adult and immature (emerged and dormant) CMM from 2015 to 2021 in burned and unburned transects.	49
Figure 24. Plots of a) CMM fruit density (fruit/m ²), b) CMM seedling density (seedlings/m ²), c) MEVE precipitation anomalies (%) (winter and spring), and d) MEVE temperature anomalies (%) (winter and spring) from 2015 to 2023.....	50
Figure 25. Distribution of CMM age class densities from 2015 to 2021 in a) burned and b) unburned transects.	53
Figure 26. Average annual values of a) pH, b) dissolved oxygen (%) and c) non-winter temperatures (°C) (March–November) from 2009 to 2021.	57
Figure 27. Average annual values of a) Nitrite (mg/L), b) Ammonia (mg/L), c) E. coli (MPN/mL), d) Iron (ug/L), e) Aluminum (ug/L), f) Cadmium (ug/L), g) Lead (ug/L), h) Manganese (ug/L), i) Molybdenum (ug/L), j) Uranium (ug/L), k) Boron (ug/L), l) Arsenic (ug/L), m) Copper (ug/L), n) Nickel (ug/L), o) Selenium (ug/L), p) Zinc (ug/L), q) Mercury (ug/L), and r) Chromium (ug/L) from 2009 to 2021.....	58
Figure 28. Streamflow trend over time at physical habitat transects at US Mancos SCPN monitoring site..	60
Figure 29. Trend in relative abundance of moderately tolerant (top) and intolerant taxa (bottom) at the SCPN upstream (US) monitoring site.	62
Figure 30. Trend in relative abundance (top) and species richness (bottom) of Trichopterans at the upstream SCPN monitoring site.....	63
Figure 31. Log-transformed relative abundance (individuals caught of species / total individuals caught) estimates for each individual species from 1944 to 2022.	66

Figures (continued)

	Page
Figure 32. Log-transformed relative abundance (individuals caught of species / total individuals caught) of bluehead sucker and speckled dace from 1944 to 2022.....	67
Figure 33. Log-transformed a) minimum flow (cubic feet per second) and b) maximum flow (cubic feet per second) reported each year at the Towaoc Gage (USGS Gage# 09371000) from 1975 to 2022.	70
Figure 34. Recent fire history in MEVE.	74
Figure 35. Categorical change in dominant vegetation type (AFG: annual forbs and grasses; BGR: bare ground; LTR: litter; PFG: perennial forbs and grasses; SHR: shrub; TRE: tree.	77
Figure 36. Computed aggregate area by cover class per year (area of each pixel multiplied by fractional cover values) for shrubs (SHR), perennial forbs and grasses (PFG), litter (LTR), trees (TRE), bare ground (BGR), and annual forbs and grasses (AFG) in Mesa Verde National Park.	78
Figure 37. Percent cover of annual forbs and grasses within MEVE from 1987 to 2023.....	81
Figure 38. Distribution of non-native plant treatments from 2012 to 2023.	82
Figure 39. Total acreage of chemical and mechanical non-native plant treatments within MEVE from 2012 to 2023.	83
Figure 40. Overview map of the SCPN sampling frames and plot locations within the PJ Ecological sites in MEVE.	84
Figure 41. Mean tally of seedlings by species and size class at the Loamy (L) and Shallow Loamy (S) ecosites in MEVE during 2008–2022.....	86
Figure 42. Mean percent canopy cover at the MEVE Loamy PJ Woodland ecosite (left) and the Shallow Loamy ecosite (right) during 2008–2022.....	87
Figure 43. a) Distribution of 265 springs within and nearby MEVE and their relationship to human disturbance in the form of trails and roads.....	92
Figure 44. a) Average annual spring discharge rates (liters/second) and b) average annual pool volumes (m ³) of MEVE springs from 1984 to 2023.....	94
Figure 45. Categorical change in vegetation type (annual forbs and grasses (AFG), bare ground (BGR), litter (LTR), perennial forbs and grasses (PFG), shrub cover (SHR), and tree cover (TRE)) over three years spanning the period of record from which data were processed: 1986, 2001 (post major fires) and 2023.	142

Figures (continued)

	Page
Figure 46. Computed aggregate area by cover class, per year (area of each pixel multiplied by fractional pixel coverage values) for annual forbs and grasses (AFG), bare ground (BGR), litter (LTR), perennial forbs and grasses (PFG), shrub cover (SHR), and tree cover (TRE).	143
Figure 47. Fractional cover for each of six classes for three select years during the first year of the period of record of RAP (1986), subsequent to the Bircher and Pony fires (2001), and most recent year available (2023).....	144
Figure 48. Sen’s slope of the trend in fractional cover for each of annual forbs and grasses (AFG), bare ground (BGR), litter (LTR), perennial forbs and grasses (PFG), shrub cover (SHR), and tree cover (TRE) for three decadal periods (1990–1999, 2000–2009, 2010–2019).	145
Figure 49. Change in fractional cover of annual forbs and grasses (AFG), bare ground (BGR), litter (LTR), perennial forbs and grasses (PFG), shrub cover (SHR), and tree cover (TRE) within the perimeters of Long Mesa fire (1989).....	146
Figure 50. Change in fractional cover of annual forbs and grasses (AFG), bare ground (BGR), litter (LTR), perennial forbs and grasses (PFG), shrub cover (SHR), and tree cover (TRE) within the perimeters of Long Mesa fire (2002).....	147
Figure 51. Change in fractional cover of annual forbs and grasses (AFG), bare ground (BGR), litter (LTR), perennial forbs and grasses (PFG), shrub cover (SHR), and tree cover (TRE) within the perimeters of Chapin 5 fire (1996).	148
Figure 52. Change in fractional cover of annual forbs and grasses (AFG), bare ground (BGR), litter (LTR), perennial forbs and grasses (PFG), shrub cover (SHR), and tree cover (TRE) within the perimeters of Pony fire (2000).....	149

Tables

	Page
Table 1. The primary drivers and stressors that influence the conditions (states) of MEVE’s focal resources.	8
Table 2. MEVE passerines and hummingbirds condition ratings summary.	31
Table 3. Summary from the Colorado Natural Heritage Program dataset (Colorado Natural Heritage Program 2023) of rare plants monitored at least once in MEVE after 2000.....	38
Table 4. The percent cover of annual grasses and forbs (Allred et al. 2021) from RAP 2023 within the surveyed distribution of 11 species of rare plants in MEVE.	43
Table 5. Chapin Mesa milkvetch (CMM) condition ratings summary.	54
Table 6. Water quality condition ratings summary, based on SCPN monitoring from 2009 to 2021.	59
Table 7. Macroinvertebrate condition ratings summary, based on SCPN monitoring from 2008 to 2019.	64
Table 8. Mancos River fish and aquatic habitat condition ratings summary.	70
Table 9. Riparian and river corridor condition ratings summary.	72
Table 10. Average quadrat (10 m ²) and plot-scale (1500 m ²) richness of all (non-tree) plants and range (lowest and highest values) for Loamy and Sandy Loamy ecological sites in MEVE during 2008–2022.....	85
Table 11. Average plot-level percent cover by lifeform in two ecosites in MEVE during 2008–2022.....	85
Table 12. Piñon-juniper habitat condition ratings summary.	89
Table 13. Condition Rating Table for avian species in MEVE.....	122
Table 14. Condition Rating Table for Chapin Mesa milkvetch in MEVE.....	123
Table 15. Condition Rating Table for Chapin Mesa milkvetch in MEVE.....	124
Table 16. Condition Rating Table for water quality in MEVE.	124
Table 17. Condition Rating Table for macroinvertebrates in MEVE.	125
Table 18. Condition Rating Table for fish and fish/aquatic habitat in Mancos River in MEVE.	126
Table 19. Condition Rating Table for riparian river corridor, using the 2024 Rapid Stream Riparian Assessment (RSRA) functional category scores, on MEVE’s stretch of the Mancos.	126

Tables (continued)

	Page
Table 20. Condition Rating Table for PJ woodland condition, using SCPN's indicators for trend assessment, PJ Loamy soil site plot data for 2007 to 2023.....	127
Table 21. AOS 4-letter codes and associated common and scientific bird names.	136
Table 22. Population trends of bird species in MEVE tracked by the Bird Conservancy of the Rockies (BCR) from 2008 to 2022, where a population mean trend) of 1 indicates the population is stable, < 1 indicates the population is declining, and > 1 is an increasing population.	138

Appendices

	Page
Appendix A. Condition Rating Tables.....	122
Appendix B. Extended Methodologies	128
Appendix C. American Ornithological Society Four-letter codes and Associated Common and Scientific Names for all Avian Species Included in NRCA Report.....	136
Appendix D. Population Trends of 53 Birds in MEVE Tracked by BCR, 2008–2022.	138
Appendix E. Quantification of Vegetation Cover Change in MEVE with the Rangeland Analysis Platform.....	141

Abstract

In 2023, the National Park Service (NPS) initiated a Natural Resource Condition Assessment (NRCA) project at Mesa Verde National Park (MEVE). The purpose of an NRCA is to: 1) synthesize information related to the primary drivers and stressors affecting natural resources identified by park managers, 2) report conditions for these natural resources, and 3) provide management recommendations to park managers based on NRCA findings. During a scoping meeting with report writers and Subject Matter Experts in September 2023, participants identified the top three drivers of change within MEVE to be climate change, increased incidence of fire, and spread of non-native plants, each of which are having direct influences over the conditions of MEVE's focal resources. Also at the Scoping Meeting, MEVE Park staff selected six focal resources for the NRCA: avifauna, deer & elk, rare plants, the Mancos River, upland vegetation, and seeps & springs. Depending on data availability, resource conditions were evaluated through either condition assessments, which assess the current status of a focal resource, or gap analyses, which summarize known information and identify knowledge gaps.

For the condition assessments, we evaluated a subset of wildlife including passerines and hummingbirds, selected rare plants, the Mancos River (specifically water quality and quantity, fish populations, and riparian corridor function), and piñon-juniper woodlands. Gap analyses were conducted for raptors, additional rare plants, deer & elk, upland vegetation (excluding piñon-juniper woodlands), and seeps & springs. Resources assessed via a condition assessment were rated using individualized condition rating tables. Additionally, report writers provided management recommendations in Chapter 4 for each natural resource.

Executive Summary

The National Park Service (NPS) Natural Resource Condition Assessment (NRCA) Program initiated an NRCA project with Mesa Verde National Park (MEVE) in 2023. The purpose of an NRCA is to: 1) synthesize information related to the primary drivers and stressors affecting natural resources identified by park managers, 2) report conditions for these natural resources, and 3) provide management recommendations to park managers based on NRCA findings. For MEVE's NRCA, managers identified raptors, passerines, hummingbirds, mule deer (*Odocoileus hemionus*) and elk (*Cervus elaphus*), rare plants, the section of the Mancos River through the Park, upland vegetation, and seeps and springs as focal resources. Resource conditions were evaluated as either a condition assessment (assessing the current status of a focal resource in the park) or a gap analysis (summarizing known information and identifying knowledge gaps), depending on data availability. Managers identified the top three drivers of change within MEVE to be climate change, increased incidence of fire, and spread of non-native plants, each of which are having direct influences over the conditions of MEVE's focal resources.

Ancestral Pueblo people have lived in and around MEVE since 550 CE. Established in 1906 by President Theodore Roosevelt, MEVE was the first national park preserved for its cultural values. But the ecological values of MEVE that attracted Ancestral Pueblo people remain important for floral and faunal species today. Over 70 species of mammals, 200 species of birds, and 1000 species of insects and other invertebrates can be found in the Park. The varied geology of MEVE translates to relatively high biodiversity of plants, including 640 plant species and four known endemic species. The upland plant communities in MEVE vary by elevation, and include sagebrush (*Artemisia* spp.), mountain shrub, piñon pine (*Pinus edulis*) and juniper (*Juniperus osteosperma*) (PJ) woodland, Douglas fir (*Pseudotsuga menziesii*) and ponderosa pine (*Pinus ponderosa*) communities. The old growth PJ woodlands still found in the Park harbor high biodiversity and are considered irreplaceable by the researchers who have studied them. Along the Mancos River and in the seeps and springs scattered across this otherwise arid landscape, unique and vital microclimates can be found in the riparian areas along riverbanks and around larger springs.

Due to the variation in habitat types, the Park supports a high diversity of avian species, according to a long-term dataset provided by MEVE, the Hummingbird Monitoring Network, the NPS Southern Colorado Plateau Network, and the Bird Conservancy of the Rockies. However, contemporary stressors are driving changes within MEVE's avifauna. It is apparent that Mexican spotted owls (*Strix occidentalis lucida*) have been extirpated from the Park. However, the gap analysis for golden eagles (*Aquila chrysaetos*) and peregrine falcons (*Falco peregrinus*) indicated that populations for these two MEVE raptors are likely stable, and in line with current trends for these species in the region. We found the condition of MEVE hummingbirds to be good/fair as trends of two non-breeding species (black-chinned, *Archilochus alexandri*, and the broad-tailed, *Selasphorus platycercus*, hummingbirds) found in the park are stable, despite some fluctuations over time. The condition assessment for passerines in the Park indicated relative stability, though our results exhibited some subtle nuances. For example, from 2008 to 2022, 11 of 53 passerine species showed

annual rates of population size increase of 10% or more per year, 12 of 53 species indicated annual rates of population size decrease of 10% or more per year, and the rest were stable. However, an analysis of the response of some of the Park's most common bird species to fire-induced habitat change indicated interesting trends: five out of 14 (36%) passerine species showed statistically supported decreases in density in burned habitat, though six out of 14 (43%) increased in density as time-since-fire increased. In light of these findings, a comparison to the Bird Conservancy of the Rockies regional dataset indicated that the occupancy rate for a majority (30) of 40 avian species analyzed is higher than the regional average.

MEVE's elk and deer gap analysis used herd estimates provided by Colorado Parks and Wildlife (CPW) to determine general trends for herds that use MEVE. Over the past two decades there has been a slight upward trend in herd population estimates for both elk and deer in management units that overlap the Park. Additionally, an overlay of regional elk and mule deer migration corridors and winter and summer ranges provided by CPW indicates that MEVE may provide an important connection between winter range and summer range to the north.

Currently, there are fourteen rare native plants of conservation concern in MEVE. The gap analysis for rare plants utilizes Colorado Natural Heritage Program (CNHP) information on surveyed known locations of these plants within the Park, the estimated density of plants within surveyed areas, and CNHP's "Element Occurrence ranks" estimating the degree of health of those populations. Additional survey data for the Cliff Palace Milkvetch (*Astragalus deterior*) allowed a cursory estimation of change over time; a comparison of 2021–2022 survey data to 2003–2007 data indicated a dramatic decline across both burned and unburned habitats within MEVE. The condition assessment for the Chapin Mesa milkvetch (*Astragalus schmollii*) revealed very different ratings depending on whether it was in burned or unburned habitats. Although fruit production and seedling survival were rated good in both unburned and burned habitats, population trends, demography and seedling density were good in unburned habitats but ranged from poor to fair/poor in burned habitats.

The condition assessment for MEVE's stretch of the Mancos River incorporated separate analyses for water quality, macroinvertebrates, fish and aquatic habitat, and the riparian corridor. The condition assessment for water quality utilized the findings of the Southern Colorado Plateau Network's (SCPN) 13-year monitoring dataset (2009–2021) within the Park's stretch of the river. SCPN's findings for both inorganic/non-mineral and nutrient indicators reflected good water quality conditions over the monitoring period, as these indicators met or exceeded state standards. On the other hand, trace metal and mineral indicators rated fair, as only 73% of these indicators met or exceeded state standards during the monitoring period, likely due in part to the effects of upstream legacy copper mining. The condition assessment for macroinvertebrates, which convey key information on how water quality and quantity affects aquatic life, were tiered to a 12-year SCPN monitoring dataset (2008–2019). Condition ratings for flow trend, relative abundance of pollution tolerant versus intolerant taxa, caddisfly trend, and percentage of samples that meet state thresholds for attainment all ranged from fair to poor, likely due to warming and drying trends in the Mancos River. The condition assessment for fish and aquatic habitat utilized a long-term CPW survey dataset, results from past Rapid Stream-Riparian Assessments (RSRA) on MEVE's stretch of the

river, and data from the closest stream gage on the river. These condition ratings, similar to those for macroinvertebrates, ranged from fair to poor, again likely due to warming and drying trends on the river. The condition assessment of the riparian corridor of the Mancos River through the Park utilized three separate RSRA assessments conducted between 2006 and 2024. Riparian vegetation health and function was determined to be in good condition, fluvial geomorphology of the corridor was found to be in good/fair condition, and riparian structure and wildlife habitat was found to be in fair condition, principally due to lack of structure in the mid and upper canopy layers of the corridor.

The gap analysis for upland vegetation in MEVE utilized the Rangeland Analysis Platform (RAP), which is a dataset modeled from long-term Landsat collection imagery. The analysis of RAP data revealed a general trend of declining tree cover and increasing shrub cover in MEVE from the mid-1980s to the present. The gap analysis for upland vegetation also considered the effects of fire on MEVE's upland vegetation communities, primarily through the summarization of the Park's Long-term Post-fire Monitoring reports, and recent research conducted by the U.S Geological Survey and Northern Arizona University. The gap analysis for upland vegetation also examined changes in non-native plant cover in the Park over time, primarily through the RAP dataset along with GIS data provided by MEVE and information in the Invasive Plant Monitoring Plan reports (IPMP). The percent cover of annual forbs and grasses was higher after stand-replacing fires than pre-fire levels. Overall, post-fire invasion of grasses coupled with climate change are creating conditions seemingly inhospitable for PJ regeneration, possibly forcing a state change from PJ woodlands to one dominated by perennial grasses and shrubs.

The condition assessment for PJ woodlands utilized SCPN's long-term (2008–2022) dataset and subsequent analyses from monitored vegetation and soils in unburned PJ woodlands in MEVE. SCPN's results were stratified according to the two soil types (loamy and shallow loamy) in which monitoring plots were located. In general, PJ indicators ranked well in the condition assessment: across the two soil types, ratings for shrub cover, perennial grass cover, annual grass cover, species richness, and PJ canopy cover all ranged from good/fair to good. Exceptions were forb cover, which ranked as good/fair in the Loamy PJ Ecosite, and fair in the Shallow Loamy PJ Ecosite; and piñon seedling recruitment, which ranked as fair in the Loamy PJ Ecosite, and fair/poor in the Shallow Loamy PJ Ecosite. This last finding is potentially concerning, as all SCPN monitoring plots were in unburned sites, portending potential future problems with piñon recruitment in MEVE even in the absence of fire.

Lastly, the gap analysis for seeps and springs in the Park utilized a long-term dataset provided by MEVE that included pool area, spring discharge, and feral livestock presence/trespass. While there is substantial year-to-year variation in both average spring discharge and average pool volume, there was not a clear trend in either metric across the period over which seeps and springs were monitored. In regard to presence or absence of feral or trespass livestock, 15% of features monitored within the past five years recorded a trespass in the most recent survey.

Acknowledgments

We are grateful to all who provided peer review of this document and/or participated in MEVE's NRCA study plan scoping meeting, which was held at the Park and virtually on September 27, 2023 (participants listed below). Also, a sincere thank you to Paul Morey (Chief of Natural Resources, Mesa Verde National Park/Yucca House National Monument), for the Park tour and scoping meeting assistance. We also thank Rachel Crafford, Western Colorado University, Candidate, M.E.M. and M.S. in Ecology, who joined the MEVE NRCA Core Team in partial fulfillment of the requirements for her M.E.M. The NPS—Natural Resource Condition Assessment Program provided funding for this project.

1. Birch, Andrew, NPS—Water Resource Division, Biologist, Water Quality & Macros
2. Brown, Nathan, NPS—Mesa Verde National Park, Wildlife Program Manager
3. Cook, Tammy, NPS—NRCA Program, Program Manager
4. Cammack, Daniel, Colorado Parks and Wildlife, Aquatic Biologist
5. Collins, Kayci Cook, NPS—Mesa Verde National Park, Superintendent
6. Floyd, M. Lisa, Prescott College, Emeritus Professor of Environmental Studies
7. Gehring, Catherine, Northern Arizona University—Department of Biological Sciences
8. Gross, John, NPS—Climate Change Response Program, Climate Science and Adaptation Coordinator
9. Johnson, Matt, NAU—Colo Plateau Research Station, Field Team Leader
10. Johnson, Matthew, NPS—Southern Colorado Plateau I&M Network, Program Manager
11. Jones, Harry, NPS—So. Colorado Plateau I&M Network and Institute for Bird Populations, Biologist
12. Ketchum, Farley Sr., Ute Mountain Ute Tribe—Traditional Ecological Knowledge Steward
13. Lutmerding, Mikey, USGS—North American Breeding Bird Survey, Biologist
14. McCauley, Ross, Fort Lewis College, Chair, Biology Department
15. Prasser, Nikita, NPS—Mesa Verde National Park, Vegetation Ecologist
16. Mack, John, NPS—Intermountain Region, Biological Resources Program Manager
17. Reed, Sasha, USGS—Southwest Biological Science Center, Research Ecologist
18. Robinson, Benjamin, Ute Mountain Ute Tribe—Biologist
19. Rondeau, Renee, Colorado Natural Heritage Program, Conservation Planner and Ecologist
20. San Miguel, George, Retired NPS—former MEVE Natural Resources Program Manager
21. Schook, Derek, NPS—WRD Water Resource Division, Hydrologist

22. Scott, Michael, USGS—retired, Riparian Ecologist
23. Spear, Andrew, NPS—Mesa Verde National Park, Physical Science Program Manager
24. Spector, Tova, USFS Intermountain Region 4, Regional Botany Program Manager
25. Stacey, Peter, retired, University of New Mexico, Riparian Ecologist
26. Stanley, Randy, NPS—IMR Natural Resources, Project Manager and IM Region Coordinator
27. Stumpf, Stacy, NPS—Southern Colorado Plateau I&M Network, Biologist
28. Swan, Megan, NPS—Southern Colorado Plateau I&M Network, Ecologist
29. Trammell, Melissa, NPS—Regional Fisheries Biologist
30. Trimber, Gillian, Northern Arizona University, PhD Candidate, Department of Biological Sciences
31. Walker, TC, Bird Conservancy of the Rockies, Staff Biologist
32. Weinmeister, Brad, Colorado Parks and Wildlife, Wildlife Biologist
33. Wethington, Susan, Hummingbird Monitoring Network, Coordinator
34. White, Jim, Colorado Parks and Wildlife, Aquatic Biologist
35. Wolcott, Sensa, Mancos Conservation District, Watershed Coordinator

Datasets used in this report were provided by the following agencies and organizations:

1. Bird Conservancy of the Rockies (avian point counts)
2. Colorado Natural Heritage Program (rare plants)
3. Colorado Parks and Wildlife (fish survey data, deer and elk herd estimates, ungulate core and corridor coverages)
4. Hummingbird Monitoring Network (hummingbird banding dataset)
5. NPS Intermountain Region (fish survey data)
6. NPS—Southern Colorado Plateau Monitoring Network (avian point counts, water quality, macroinvertebrates, PJ woodland vegetation monitoring data)

Glossary

Condition Assessment: A condition assessment reports on the current condition of one or more *indicators of condition* for a focal natural resource when enough data are available. What constitutes “enough” data is a professional judgment based on a review of all available data and discussions with park staff, subject matter experts, and researchers involved in collecting the applicable data. Indicators of condition are rated using a three- or five-level rating system, ranging between good-fair-poor. If trend can be reported for a measure, it is reported as improving, stable, deteriorating, or unknown/indeterminate.

Condition Evaluation: A condition evaluation is either a condition assessment or a gap analysis. Which is conducted for a given resource depends on the amount of data with which to evaluate conditions and assign a condition rating(s). If enough data are available for key indicators of condition, then a condition assessment is developed. If data are lacking, a gap analysis is developed.

Condition Rating Statements: These correspond to the three- or five-level rating system developed to evaluate the indicators of condition. The five-level rating scheme includes good, good/fair, fair, fair/poor, and poor, with corresponding “stop light” colors. Condition ratings statements are developed at the indicator level for the combination of measures evaluated for each indicator of condition. The condition rating statements’ reference criteria must be logical and defensible based on the best available science. In the case of the MEVE condition rating tables, they were built using a combination of best available science (references to published scientific literature), expert opinion, and, where available, field data collected within MEVE.

Confidence Level: These correspond to a three-level rating system of low–medium–high described for the indicators of condition ratings. The levels are based on the repeatability of evaluation findings and how confident the authors are in the information used to rate condition.

Current Condition: This defines the status of condition for an indicator based on the evaluation of one or more measures. “Current” applies to the condition as it exists today based on what has previously occurred, not on what is likely to occur. For example, something such as hazard level or risk, which identifies the likelihood of what may occur because of the intrinsic characteristics of the resource, will not be used to report on current condition. In general, data collected within the last ten years can be used to determine current condition.

Data: Descriptive or numeric information about an object, group, process, or phenomenon. There are three kinds of data typically used in natural resource analyses- ordinate (ordered, as in poor, fair, good), categorical (discrete values that fall into distinct categories), and continuous (any value within a defined range, often on a continuous scale).

Data Gap: A data gap occurs when information is lacking, whether in the form of insufficient literature or subject matter expertise to adequately evaluate conditions.

Driver: Ecosystem drivers are major (and most often) external influencers of change to natural systems, functioning across extensive areas or scales. Drivers are defined as “any relatively discrete events in space and time that disrupt ecosystem, community, or population structure and change resources, substrate, or the physical environment” (Pickett et al. 1985). Drivers are most often beyond a manager’s ability to influence or change.

Gap Analysis: A gap analysis summarizes what is known about a focal natural resource, in addition to highlighting critical information that is lacking. A gap analysis does not rate indicators of condition. Instead, a table of *proposed* indicators, measures, and reference criteria are reported, with the goal of providing a framework for a future study.

Indicator of Condition: An indicator of condition (or simply indicator) is a descriptor of something useful to measure, but it is not the measure itself. Indicators consist of one or more measures. Condition ratings are assigned at the indicator level. This is because natural resources are often more complex and nuanced than what is reflected in just a few measures and associated indicators.

Measure: A measure is qualitative, quantitative, or a combination of both and provides specific information about the indicator of condition. There can be one or more measure(s) for each indicator. Selected indicators and measures are often those that are commonly used by NPS staff in monitoring the status of a resource, as well as those that are well represented in the literature and can provide context when park-specific data are lacking.

Pressure: A pressure results from a driver, potentially affecting a resource. An NRCA presents drivers and pressures as the fundamental forces that play important roles in regulating or altering ecological resource conditions in the Park. NRCAs do not differentiate between drivers and pressures because the focus of the report content is on the manifestation of those influences on natural resource conditions, not on the differentiation of drivers and pressures.

Reference Criteria: Reference criteria are pivot points, thresholds, or ranges based on peer-reviewed literature, state standards, known criteria, or some other justifiable source of information that forms the basis of the condition rating statements. Quantitative reference criteria are generally better than qualitative reference criteria, but when specific data are lacking, qualitative reference criteria are useful. Regardless of the type of reference criteria used, they must be justifiable and cited appropriately for the repeatability of future condition evaluations.

Stressor: “Stressors can manifest as physical, chemical, or biological perturbations [disturbances] to a system that are either foreign to that system, or natural to the system, but occurring at an excessive or deficient level. Stressors cause significant changes in the ecological components, patterns, and processes in natural systems. They act together with drivers on ecosystem attributes” (Barrett et al. 1976). When possible, stressors are selected as measures with which to evaluate the current condition of an indicator.

Trend: A trend is a statistical analysis intended to find patterns in data. If a trend can be reported for a measure, it is reported as improving, stable, deteriorating, or unknown/indeterminate. Only quantitative trends are reported in the NRCA technical report.

Natural Resource Condition Assessment—Introduction

The National Park Service (NPS) Natural Resource Stewardship and Science Directorate provides scientific, technical, and administrative support to national parks for the management of natural resources. The directorate includes nine Service-wide divisions that assist NPS managers across the United States with protecting park resources and values sustainably over time. The NPS Natural Resource Condition Assessment (NRCA) Program is within the Service-wide's Inventory and Monitoring Division, and its mission is to assess and report the conditions and trends of natural resources.

Conceptual Framework for Assessing Resource Conditions and Influences

NRCA projects employ a Drivers, Pressures, Stressors, States, and Responses (DPSSR) conceptual framework (Figure 1) similar to that used and advocated by others (Harwell et al. 2019). The DPSSR framework structures the content in an NRCA report and guides discussions of the anthropogenic and/or natural processes that “drive” ecosystem change. While drivers or pressures (hereafter, drivers) may affect the current condition (or state) of a natural resource, it's important to note that drivers related to natural processes do not always act as stressors. However, some historically natural processes that drive change in these landscapes may become stressors due to anthropogenic influences altering their magnitude, frequency, duration, or other attributes. For stressors that are within the Park's “span of control”, a Park management response may maintain, improve, or restore the natural resource indicator of condition to a desired state. For stressors that the Park cannot directly manage or influence, it's still helpful to understand potential impacts, especially when identifying realistic resource management goals.

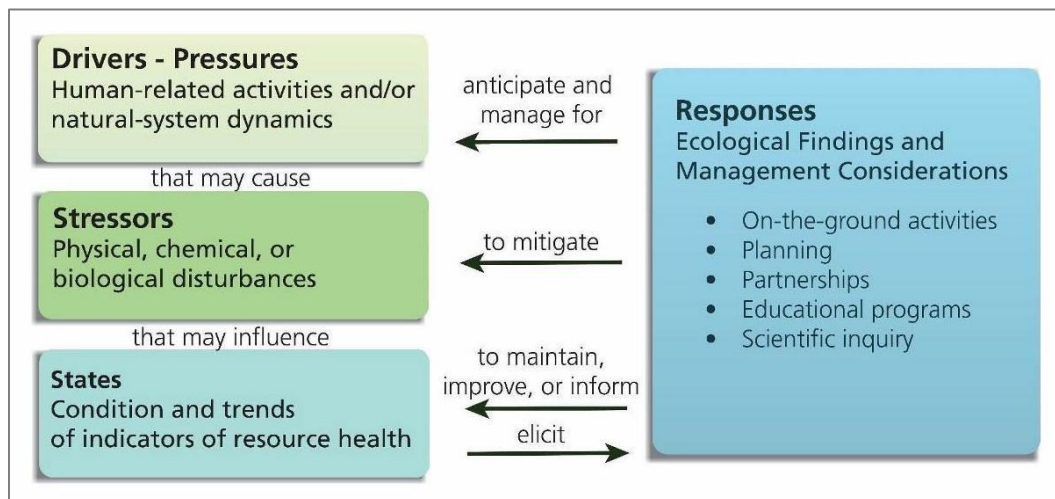


Figure 1. The drivers, pressures, stressors, states, responses (DPSSR) logic framework used to organize NRCA report content. Modified from content originally published by Harwell et al. (2019).

NPS / CONSERVATION SCIENCE PARTNERS

Even though there are stressors that managers cannot directly influence, it is still useful to understand the potential resource impacts, especially when setting realistic resource management goals. The

DPSSR framework helps to illuminate these connections between drivers, stressors, resource conditions (states), and management responses and guides the selection of the most relevant NRCA content to report.

While an NRCA project does not report on conditions for all the natural resources at a Park, the DPSSR framework can guide the evaluation of additional natural resource conditions in the future. This is especially important as drivers or stressors change or as new information becomes available with which to evaluate natural resource conditions.

NRCA Goals

The primary goal of a NRCA is to deliver scientifically credible natural resource condition and trend information for park manager-selected topics of interest. NRCAs distill existing information into concise, easily understood assessments that inform management decisions. NRCAs also highlight data gaps and potential research needs to help managers prioritize information that is needed to inform future management decisions. The timing of the Mesa Verde NP (MEVE) NRCA is particularly advantageous as this snapshot of resource conditions will serve as a helpful “baseline” to MEVE as it moves forward with development of the MEVE Resource Stewardship Strategy (RSS) in 2025.

Each NRCA project is completed within about one year from the scoping workshop with the development of the technical report, using a standard workflow. The NRCA findings are published in a technical report, which is posted to the NPS Data Store along with supporting materials (e.g., GIS datasets, study plan, assessment methods summaries) (<https://irma.nps.gov/DataStore/>).

Technical Report

The content of every NRCA report is organized using an ecological drivers, pressures, stressors, states, and responses (DPSSR) logic framework (Figure 1). The DPSSR framework emphasizes the connection between the natural and human (or anthropogenic) drivers-pressures that influence ecosystem or natural resource change. A change may affect the condition (or state) of a resource as a positive or negative stressor. Park managers may respond to a stressor(s) that negatively impacts a resource to restore its condition to a desired state, such as controlling non-native invasive plants in a high-priority habitat. As such, the NRCA report will include some high-level management recommendations, based on the conditions evaluations for the focal resources presented in the report.

Chapter 1. Introduction to Park Setting and Resources

Mesa Verde National Park (MEVE) is one of 63 national parks included in a system of 431 national park units. It was established on June 29, 1906, by President Theodore Roosevelt to preserve MEVE “*from injury or spoliation of the ruins and other [prehistoric] relics*” Mesa Verde was the 6th national park to be designated by Congress, but the first to preserve nationally significant cultural resources. Established in 1906, it remains the only national park created to protect cultural resources (National Park Service 2015). MEVE is administered by the National Park Service (NPS), one of nine bureaus and offices of the Department of Interior. NPS conserves America’s vast natural and cultural resources within all its park units for the enjoyment and use of future generations.

This land was inhabited by Ancestral Pueblo people from approximately 550 to 1300 CE (National Park Service 2015) and was designated as a UNESCO World Heritage Site on September 8th, 1978, for containing some of the best-preserved Ancestral Pueblo archaeological sites in the country (National Park Service 2015). Given the profound cultural significance of this land, it is essential to acknowledge both the history behind MEVE’s establishment and the enduring ancestral connections many Indigenous groups maintain to today. Currently, 24 federally recognized different tribes are associated with Mesa Verde, including the 19 Pueblos of present-day New Mexico, the Hopi Tribe, the Ysleta del Sur Pueblo in present-day Texas, the Navajo Nation, and the Southern Ute and Ute Mountain Ute peoples (National Park Service n.d.).

Due to the Park’s proximity to the Ute Mountain Ute Reservation and the role that the Ute Mountain Ute Tribe plays in its stewardship, it is necessary to provide context on the historical relationship between the Ute peoples and this land. By the early 1800s, Utes had been living in what is now western Colorado for over 500 years. However, by the late 19th century, American pressures to mine Ute land throughout the region prompted the Ute Treaty of 1868, which established a reservation across the western third of Colorado (History Colorado 2024). Seven bands of Ute peoples living in Colorado and Utah were relocated to this reservation, ceding ancestral lands to the United States (History Colorado 2024).

Despite MEVE being located on reservation lands deeded to the Ute Mountain Ute Tribe (UMUT) through the Ute Treaty, shortly after the Park’s establishment the U.S. government proposed a land exchange, offering compensation in return for incorporating ancestral lands and cultural features from the Reservation into the Park boundary (Smith 2002). In spite of significant opposition from Tribal members, the exchange proceeded, granting the Tribe the northern half of Sleeping Ute Mountain. However, many Tribal members viewed this land as sacred and already rightfully theirs, making the compensation deeply contentious (Smith 2002). This historical context, along with perspectives and insights from Ute peoples, is fundamental to understanding MEVE’s story and should guide efforts to honor the land’s complex history and cultural legacy moving forward.

1.1 Location

MEVE is located in southwestern Colorado near the Four Corners region, where the states of Colorado, Utah, Arizona, and New Mexico meet. The entrance to the Park is along Highway 160

approximately 13 km (8 mi) west of Mancos CO., and 16 km (10 mi) east of Cortez CO (Figure 2). The Park's legislative boundary encompasses approximately 21,044 hectares (52,000 acres) of the Colorado Plateau in Montezuma County.

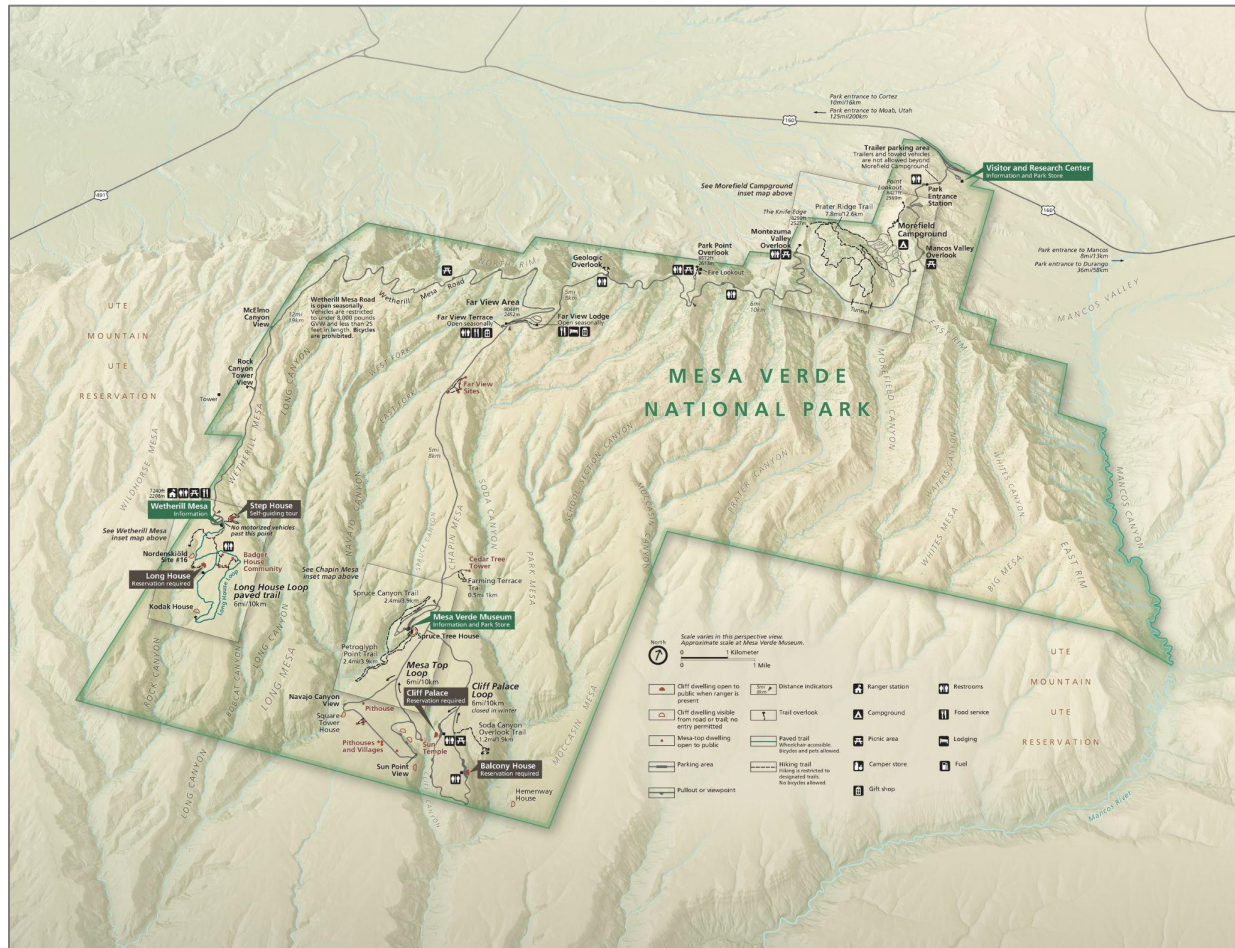


Figure 2. NPS map of MEVE and the surrounding region.

NPS



Sun setting near the Knife Edge at Mesa Verde National Park.

NPS / JACOB W. FRANK

1.2 MEVE Purpose, Significance and Fundamental Resources and Values

MEVE’s purpose statement from the 2015 Foundation Document is: “Mesa Verde National Park protects, preserves, researches, and interprets an archeological landscape including more than 600 cliff dwellings, wilderness values, and remarkable scenic resources in southwest Colorado” (National Park Service 2015). More than 5,000 known archeological sites lie within MEVE’s boundaries, including not only cliff dwellings but also pithouses, pueblos, masonry towers, and farming structures (National Park Service 2015). These special sites document the culture of the Ancestral Pueblo people who lived on the mesa for more than 700 years and offer insights into their lives, as well as this chapter of human history on the Colorado Plateau (National Park Service 2015).

The scenic resources of MEVE are considered to be one of the “Fundamental Resources and Values” of the Park. Sixteen percent of MEVE is designated wilderness, with three separate wilderness areas totaling nearly 8,000 acres. These areas are largely undeveloped with no maintained trails, campsites, or structures (National Park Service 2015). The viewshed of MEVE includes this wilderness, along with mountains, nearby and distant canyons, escarpments, and valley floors, which is the same landscape seen by the Ancestral Pueblo people (National Park Service 2015). For example, visitors of MEVE today can experience a night sky very similar to what the Ancestral Pueblo saw a millennium ago (Olstad 2024b). MEVE’s landscape retains evidence of habitation, defense, farming, and ceremony, and provides a physical, historical, and spiritual context for the Park (National Park Service 2015).

While the Park was originally created to chiefly protect archeological sites and promote scenic values, the ecological values of MEVE that attracted Ancestral Pueblo people remain important for

fish and wildlife species today. Two-needle piñon pine (*Pinus edulis*) and Utah juniper (*Juniperus osteosperma*) (PJ) woodland is perhaps the best example of these culturally significant habitat types, and many Indigenous cultures, such as the Hopi, Diné, and Ute peoples, continue to rely on both piñon and juniper trees as food sources, for medicine, and in celebration (Zafar 2019). To this day, the vegetation types that characterize MEVE are deeply intertwined in the cultural fabric of the landscape.

1.3 Climate

Warm-to-hot summers and cold-to-mild winters characterize MEVE's semi-arid climate (MEVE NP Colorado 2024). From 1925 to 2023, the warmest temperatures occurred in July and August, with an average of 72°F and 70°F respectively, and the average coldest temperatures occurred in January (29°F). Precipitation has been highest during August, with an average of 5.05 cm (1.99 in) (1925–2023). Warming temperatures have driven a higher prevalence of droughts and intense wildfires (MEVE NP Colorado 2024). During July 2024, Montezuma County (where MEVE is located within CO) was not in a drought (National Drought Mitigation Center n.d.). However, between 2018 and 2024, Montezuma county had experienced 2 instances of exceptional drought (the highest rated drought a location could experience according to the U.S. Drought Monitor) (National Drought Mitigation Center n.d.).

1.4 Unique Ecological Qualities of Mesa Verde

Like elsewhere in the Four-corners region of the Colorado Plateau, the flora, fauna and key resources of MEVE are strongly tied to its semi-arid climate, combined with its geology and topography (MEVE NP Colorado 2024). The relatively low average annual rainfall (45.34 cm [17.85 in]) favors organisms adapted to drought. The varied geology, including sweeping cuestas and distinct geological formations exposed at different elevations, translates to a relatively high biodiversity of plants, including 640 plant species and four endemics known to the Park (Colorado Natural Heritage Program 2023).

MEVE is in the transition zone between the arid shrublands to the south and the forested environments of the Rocky Mountains to the north (National Park Service 2015). Elevations in the Park range from about 6,000 feet in the canyon bottoms near the southern Park boundary to 8,572 feet at Park Point, about 10 miles north. The upland plant communities in MEVE vary by elevation and aspect, and include sagebrush, mountain shrub, PJ woodlands, and Douglas fir (*Pseudotsuga menziesii*) and ponderosa pine (*Pinus ponderosa*) communities. The importance of past and current fire regimes and cycles that have shaped these communities cannot be understated; persistent PJ woodlands have been thought to sustain infrequent, high-severity crown fires with a return-interval of anywhere from 200 to 400 years (Floyd et al. 2004; Thomas et al. 2009; Turner et al. 2007). But fires are becoming more frequent in this part of the Colorado Plateau today and are leading to slow transition of PJ woodlands to mountain shrub (Phillips et al. 2023). The old-growth PJ woodlands still found in the Park harbor high biodiversity, and are considered irreplaceable by the researchers who have studied them (Floyd 2003).

The wildlife of MEVE are, like the Park's vegetative communities, adapted to the arid climate, but tend to have somewhat wider ranges than the Park's plants because they can move to seasonal water sources or to lower elevations or latitudes in the winter. Over 70 species of mammals (including many species of bats), 200 species of birds, and 1,000 species of insects and other invertebrates have been recorded in the Park (Colorado Natural Heritage Program 2023). Notable species include, golden eagles, peregrine falcons, Mexican spotted owls, four species of hummingbirds, bobcats, coyotes, elk, deer, mountain lions, rabbits, weasels, lizards, and snakes.

One reason MEVE has high wildlife diversity is that this remote part of the Colorado Plateau provides a critical refuge for wildlife. Because the Park is only accessible via U.S. Route 160 to the north and main park road is not a through route, much of MEVE is distant from human contact. MEVE's remote setting translates to other important values for wildlife such as a very quiet, natural acoustic setting and dark night skies. While a lack of sound and light pollution are arguably important for virtually all wildlife, large expanses of dark night skies are particularly important to some migratory birds, which navigate by the stars. A recent condition summary authored by NPS of MEVE's night skies found that the Park's skies rated between good and excellent using five different measurement techniques and standards (Olstad 2024b). A paired condition summary of the acoustic environment at MEVE, also authored by NPS, found that the Park currently preserves a mostly intact acoustic environment (Olstad 2024a).

There are additional non-elevation dependent ecosystems in MEVE, and these are tied to the hydrological systems of Mancos River and seeps and springs that dot the otherwise arid landscape. Here one can find unique and important microclimates in the riparian systems along the riverbanks and surrounding the larger springs. Water seeps through the permeable sandstone and shale and expresses as seeps and springs, sustaining important wetland and riparian plants, as well as wildlife, especially those that rely on these oases for both annual life needs and migration (Anderson et al. 2024). Because of these unique and important ecological values, MEVE has seen a significant amount of ecological study. This ranges from long-term monitoring and research on migratory birds (Bird Conservancy of the Rockies 2024; Crafford et al. In preparation; NPS Southern Colorado Plateau I&M Network 2024a), hummingbirds (Hummingbird Monitoring Network 2024), water quality of the Mancos River (NPS Southern Colorado Plateau I&M Network 2024b), intact PJ woodlands (NPS Southern Colorado Plateau I&M Network 2024c) and wildfire recovery dynamics (Floyd et al. 2021; Phillips et al. 2023).

Chapter 2. Drivers and Stressors

Drivers and stressors include both natural processes and human related activities, and sometimes the interaction between them. While a natural process should not usually act as a stressor to a system, if it causes change outside the historic range of variation, it could then be considered a stressor. The drivers and stressors that are likely influencing MEVE's natural resource conditions were discussed among the invited subject matter experts and MEVE natural resource staff during the 2023 NRCA scoping workshop and summarized in Table 1. Currently, the most significant stressors acting upon MEVE are climate change, the spread of invasive non-native plants, and an increased incidence of fire outside of the historic fire regime. As such, these drivers and stressors receive chief focus in this chapter, including summaries of the workshop discussions related to these landscape scale drivers and stressors, combined with a brief and pointed literature review. The more localized, resource-specific stressors are presented in Chapters 3 and 4.

Table 1. The primary drivers and stressors that influence the conditions (states) of MEVE's focal resources. The categories are from the *NRSS Key Issues, Stressors, & Threats (KIST) Menu for Park Resource Stewardship Strategies* (National Park Service 2022).

Drivers <i>Fundamental Forces to Consider</i>	Stressors <i>Potential Influences to Consider</i>
Fire and Fire Suppression	Habitat fragmentation and effects on population isolation, loss of species
	Habitat loss or conversion
	Runoff, sedimentation, contamination of Mancos River
	Departure of historical fire-cycle frequency
(Mancos River only) Upstream uses and impacts to river/watershed (grazing, agriculture, etc.)	Water pollution (metal and nutrient inputs, etc.)
	Sediment inputs (from upstream erosion)
(Mancos River only) Upstream dams and diversions	Water depletion
	Alteration of hydrograph
	Loss of beavers in system
	Sedimentation
Major roads/road use	Habitat fragmentation
	Stress and/or injury to terrestrial animals, including birds
Invasive & Problematic Species	Chiefly non-native invasive plants, may cause habitat alteration and degradation for native plant community, alteration of fire frequency.
Climate Change	Changes in temperature and precipitation regimes (and associated droughts)
	Depletion of groundwater
	Dust on snow (increased dust in snowpack from increased erosion in region leads to quicker snowmelt)
	Drought-exacerbated bark beetle (<i>Ips confusus</i>) outbreaks

Table 1 (continued). The primary drivers and stressors that influence the conditions (states) of MEVE’s focal resources. The categories are from the *NRSS Key Issues, Stressors, & Threats (KIST) Menu for Park Resource Stewardship Strategies* (National Park Service 2022).

Drivers <i>Fundamental Forces to Consider</i>	Stressors <i>Potential Influences to Consider</i>
Feral/trespass livestock	Microscale habitat alteration, including legacy effects, competition with native wildlife.
Public access and the infrastructure that supports it	Introduction of invasives
	Alteration of habitat
	Fragmentation of habitat and management decisions that can feed back into other stressors
	Loss of sensitive plants
Surrounding land uses	Habitat fragmentation
	Roads impacting migration corridors and hydrology
	Non-native species (domestic sheep, plants)
	Water withdrawals
	Dust on snow

2.1 Climate Change

Of all the drivers and stressors acting on MEVE resources, climate change is likely the most profound and ubiquitous, whether “acting alone” as a stressor or exacerbating other stressors, such as contributing to the alteration of the natural fire regime or encouraging proliferation of non-native invasive plants. For example, a severe and ongoing drought in this part of the Colorado Plateau started in approximately 1995 and, in combination with fuel accumulation during two wet decades before 1995, resulted in large crown fires. These fires were similar in character to previous fires but with shorter than normal intervals between them; specifically, with six large wildfires between 1996 and 2003 (Floyd et al. 2004; Thomas et al. 2009; Turner et al. 2007). The interplay between drought and other effects of climate change, such as insect infestations, is contributing to changes in the historic fire return interval on the Colorado Plateau (see below). Breshears et al. (2005) reported 90% or greater drought-induced mortality of piñon pines at some high elevation sites in Colorado (including MEVE) and Arizona, which they attributed to low soil water content and region-wide bark beetle infestations. It is well documented that the 2001 to 2003 drought supported an *Ips confusus* (piñon engraver beetle) outbreak from 2001 to 2004 in the Southwest including at MEVE, killing an estimated one-third of piñon pines (Breshears et al. 2005; Negrón and Wilson 2003). Further, Floyd et al. (2009) found that during the recent drought on the Colorado Plateau, piñon pines experienced mortality rates of 32–65% primarily attributed to *Ips confusus* across three sites (including MEVE) in three different states. Tree mortality is predicted to continue to increase and become more widespread on the Plateau due to the combined influences of drought and a warming climate (Schwinning et al. 2008).

While projections regarding future climate change in MEVE will be more fully addressed in the Resource Stewardship Strategy, a recently published Climate Change Exposure Report for MEVE

(Climate Change Response Program 2024) found that temperatures in MEVE have already increased significantly over the last century (Figure 3). The Climate Change Response Program (2024) reported an average annual temperature increase of 2.4°F per century from 1900 to 2022, and since 1970 at an increased rate of 6.5°F per century. Temperatures from 2000 to 2012 were consistently higher than the “anomalously” high temperatures recorded in 1934 and 1954. Since 1970, precipitation has decreased but remains highly variable, which is characteristic of the Colorado Plateau, though years with historically high precipitation totals occur less frequently (Climate Change Response Program 2024) (Figure 3).

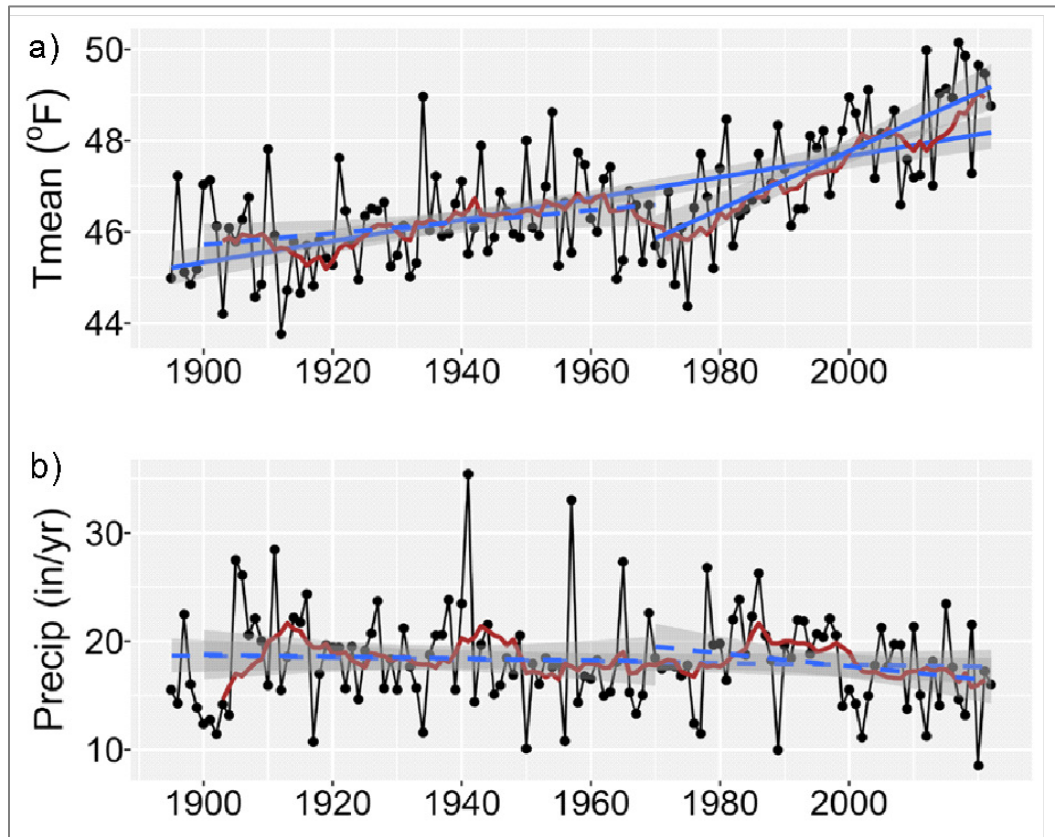


Figure 3. a) Historical trends of yearly temperatures and b) yearly precipitation from 1895 to 2022 for MEVE. Black points are annual values and the red line is the 10-year average. Blue lines are linear regressions. Solid lines represent trends that are significant ($p < 0.05$), and dashed lines are not statistically significant. Figure reproduced from Climate Change Response Program (2024).

NPS / CONSERVATION SCIENCE PARTNERS

Of particular concern is that climate change could potentially create inhospitable conditions for specific native MEVE species in the future. The recent climate change exposure report detailed plausible climate trajectories for MEVE and predicted the two possible futures for MEVE as “warm wet” or “hot dry” (National Park Service 2023). The authors described projected changes in monthly precipitation in 2050 based on these two scenarios (Figure 4a and b). Predictions include mostly

wetter winters (December–February) (Figure 4a), drier springs (March–May) (Figure 4a), and hotter winters and springs (Figure 4b).

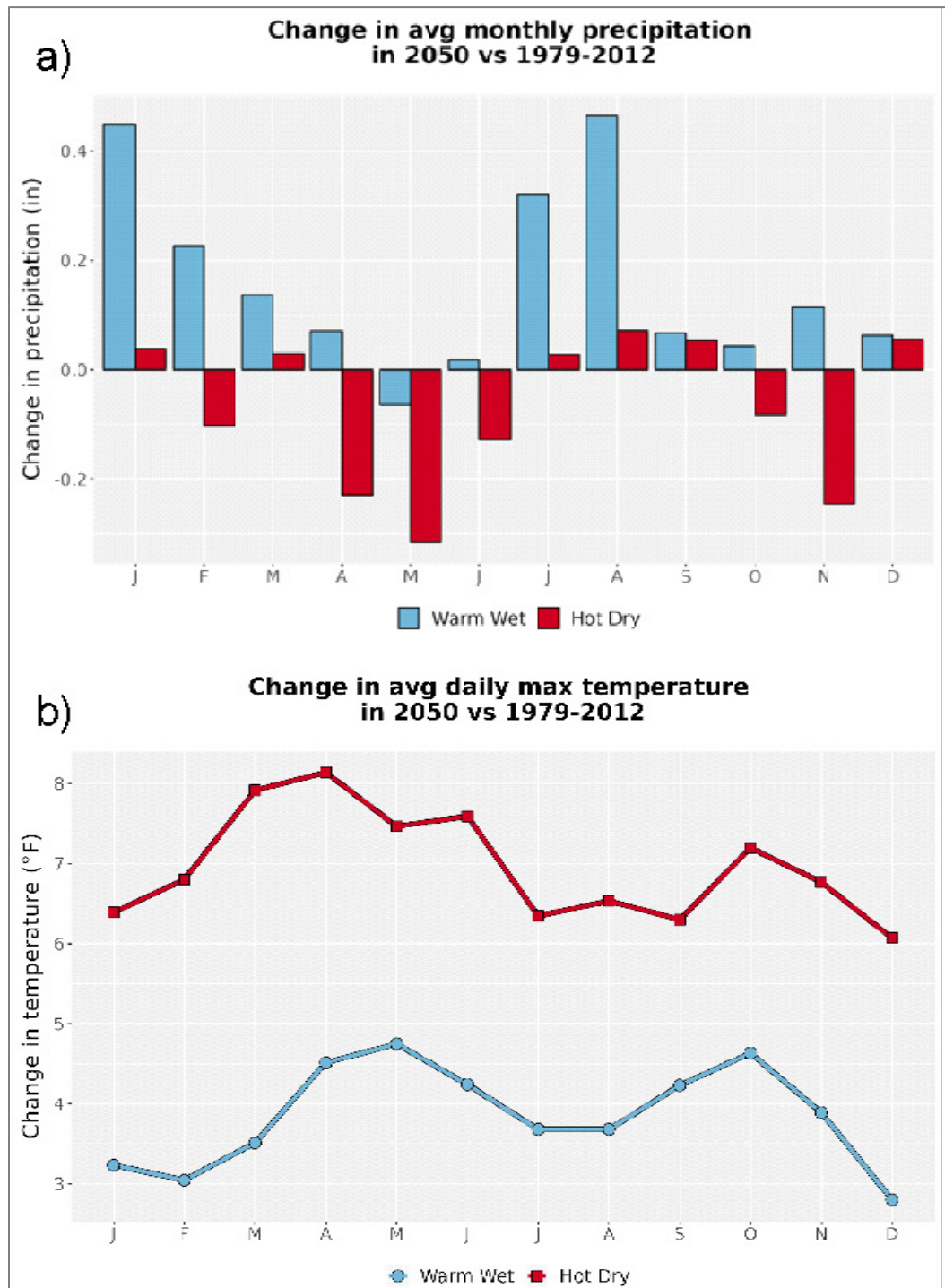


Figure 4. Change in a) monthly average precipitation and b) monthly average temperature in 2050, relative to historical conditions, based on two climate scenarios: “warm wet” and “hot dry”. Figure reproduced from National Park Service (2023).

NPS / CONSERVATION SCIENCE PARTNERS

Due to several native and endemic species in MEVE being environmentally sensitive, a potential shift towards drier and hotter conditions could impact these species. Change in timing and availability of floral resources and/or unfavorable weather conditions could also affect pollinators and limit pollination and seed production of rare plants. For example, certain flowers important to hummingbirds could senesce earlier than they usually do due to a warmer climate, which could translate to a lack of nectar resources, including during critical windows when young are fledging from nests (personal communication, Susan Wethington, Hummingbird Monitoring Network, September 2023). Another example of change likely to manifest with warming pertains to the rare Chapin Mesa milkvetch (*Astragalus schmolliae*), featured in section 3.3. While the milkvetch is a long-lived plant adapted to severe droughts and fires, the indirect effects of climate change could affect this species, as a hotter and drier climate is likely to increase the risk of old-growth PJ woodlands burning. While the milkvetch can withstand fires, it does not appear to be able to withstand a type conversion to grasslands (personal communication, Renee Rondeau, CO Natural Heritage Program, April 2024), as discussed in section 3.3. Even if the milkvetch's old-growth PJ habitat in MEVE persists despite increased chances of burning, increasingly dry conditions are likely to lead to decreased regeneration and increased mortality of PJ, especially piñon pines, in the near future (Swan 2024). In fact, it is predicted that the distribution of PJ woodlands in the West is projected to shrink with climate change (Noel et al. 2024).

2.2 Increased Incidence of Fire

Fire has played an important role in shaping both the historic and recent woodland dynamics of MEVE. Historically, persistent PJ woodlands sustained infrequent, high-severity crown fires with a return-interval of approximately 400 years (Floyd et al. 2004), while mountain shrublands on the northern portions of the mesas burned more frequently and with lower severity, with fire return intervals of about 100 years (Floyd et al. 2000b). Since the mid-1990s, drought has been frequent in the MEVE region, and as noted above, likely contributed to the unprecedented fire activity in the Park, with fully two-thirds of MEVE experiencing fire from 1996 to 2006 (Turner et al. 2007), which is well outside the natural range of variation (Floyd et al. 2004). As post-fire reorganization unfolds throughout the Park, it will be important to assess whether burned areas will recover, or if they will undergo vegetative type-conversion, shifting from one predominant habitat type to another (Guiterman et al. 2022).

Similar to trends seen within MEVE, throughout the greater Southwest across woodland and forest types, fires have become larger, more severe, and in some cases more frequent (e.g., O'Connor et al. 2020). These trends have been well documented in the ponderosa pine forests (e.g., McClure et al. 2024) and less so in the PJ woodlands of the Colorado Plateau (Reinhardt et al. 2020). Ecological repercussions of this increased fire frequency could include shifts from woodland to shrublands, as seen in southern Arizona (Barton and Poulos 2018). Floyd et al. (2004) remarked that “we may be witnessing a unique period in the ecological history of the southwest ... when vegetation patterns are being altered over extensive areas within a very short time”.

While mountain shrubland communities in the Park are showing resilience in post-fire recovery and are quick to resprout, the Park's burned Douglas fir woodlands have not recovered and likely never

will (personal communication, Tova Spector, USFS Intermountain Region 4, October 2024). Further, as we detail in Chapter 3, many PJ woodlands that have burned in MEVE are experiencing low rates of regeneration and are now dominated by shrubland or grassland states. This pattern aligns with natural post-fire succession in PJ woodlands, which was well-documented before the recent drought (Erdman 1970). Junipers are returning to burned areas on the southern portions of the 2002 fire (Floyd-Hanna, emeritus Prescott College, field observations October 2024), a trend consistent with earlier 1959 and 1934 fires in MEVE. However, given increased stressors within post-fire communities (e.g., climate change-induced drought) and a low fire-return interval in these areas, it is possible that the lack of conifer recovery may suggest the beginning of type conversion, especially in MEVE's Douglas fir woodlands.

2.3 Spread of Invasive Non-native Plants

Increases in invasive non-native plants (hereafter non-native plants) across the Colorado Plateau can represent a significant driver of change to native habitat types (Floyd 2003; Wilcox et al. 2012). These non-native species, with cheatgrass (*Bromus tectorum*), musk thistle (*Carduus nutans*), and Canada thistle (*Cirsium arvense*) serving as the principal concerns, may invade intact and undisturbed communities (Floyd 2003). Typically though, bare ground, especially that which has been recently disturbed, is most susceptible to the establishment of non-native species, offering germination sites following disturbances ranging from fire to vegetation clearing for human activities (Belsky and Gelbard 2000; Floyd 2003). Some species, such as Russian thistle (*Salsola* spp.) and cheatgrass, have slowly been expanding across the Plateau over the past 150 years, a process exacerbated by surface disturbance and removal of bunchgrasses by livestock grazing (Schwinning et al. 2008). It is important to note that, while non-native plants can be impactful, currently their effects are not as far-reaching in MEVE as other stressors discussed in this report. However, they are included here due to their ability to drive change at the local level, and their possible long-term impacts if left unmonitored.

Cheatgrass is considered one of the prominent non-native species on the Colorado Plateau (Floyd et al. 2003; Wilcox et al. 2012). When established, cheatgrass is known to outcompete native forbs and grasses (Bates et al. 2007; Molvar et al. 2024 and references therein), and has been shown to alter fire regimes at high abundance due to its flammability (Mack 1986; Molvar et al. 2024; Wilcox et al. 2012). While cheatgrass is a matter of concern in MEVE, it is not currently a major problem in terms of overall acreage affected (Allred et al. 2021). However, it needs to be monitored closely given its potential to change future fire behavior in ways that could threaten the recovery of recently burned PJ stands (Floyd 2003) since it is one of the first species to invade burned systems after a fire (Molvar et al. 2024).

Other non-native species currently in MEVE and surrounding areas include musk thistle, prickly wild lettuce (*Lactuca serriola*) and yellow alyssum (*Alyssum alyssoides*), all of which notably increased in this part of the Colorado Plateau in the 1970s and 1980s (Floyd 2003). These three species and cheatgrass increased after the fires that occurred in MEVE in 1989, 1996 and 2000 (Floyd 2003). In addition, Russian knapweed (*Rhaponticum repens*) has invaded the Mancos Canyon and is found in

the same habitat as some of the Park's endemic species, such as *Gila haydenii* and *Lepidium crenatum* (personal communication, Tova Spector, USFS Intermountain Region 4, October 2024).

Chapter 3. Focal Resource Evaluations

This chapter reports on the states (conditions) of MEVE’s selected focal natural resources. We evaluated these resources, either as a condition assessment or through gap analysis, depending on data availability.

For focal resources that lack adequate data for credible evaluations of their current condition, we developed a gap analysis. A gap analysis is a summary report on a focal resource, including its ecological importance, the general status of knowledge regarding factors influencing its condition, and the indicators, measures, and studies that may be useful for informing the Park’s understanding of the resource and management options. A gap analysis is generated through an intensive literature review combined with expert input.

A condition assessment was developed for focal resources for where adequate data on current conditions exist. For each indicator, data for one or more measures were evaluated qualitatively and/or quantitatively and that evaluation was used to report the indicator level of that resource. Rating statements include the combined measures’ qualitative characteristics and, when available, the quantitative values within either four or six condition scores, ranging from good to poor, and unknown (Figure 5). Condition scores are colorized, as shown in the figure, and when the indicator of condition is unknown it is shown in gray. Rating statements for assigning a condition level are included in Appendix A. A condition rating for the focal resource itself will not be reported because of the complexity of adequately characterizing the condition at that level. Typically, each condition rating is assigned a confidence level, reported as high, medium, or low, depending on factors such as repeatability, age of data, and whether field data were collected or analyses were model driven. However, as we are confident that the condition ratings provided in this report accurately reflect current conditions in the Park, we have high confidence levels in all condition assessments reported. If available, a statistical trend of “improving”, “stable”, “deteriorating”, or “unknown” is reported.



Figure 5. Indicators of condition rating classes and colors.

NPS

MEVE’s natural resources that were evaluated through condition assessments include: a subset of wildlife (passerines and hummingbirds); selected rare plants; the Mancos River including water quality and quantity, fishes, and riparian corridor function; and PJ woodlands. Gap analyses were developed for raptors, other rare plants, deer and elk, upland vegetation (not including PJ woodlands), and seeps and springs.

3.1. Wildlife—Avifauna (Raptors, Passerines, and Hummingbirds)

MEVE contains several distinct habitat types including PJ woodlands, Douglas fir (*Pseudotsuga menziesii*) forests, riparian landscapes, and mountain shrublands (Thomas et al. 2009). Due to this habitat diversity, many bird species breed and/or reside within the Park’s boundaries. According to

Bird Conservancy of the Rockies observations made within MEVE, 82 species were detected in the Park between 2008 and 2022, including 63 species of passerines (Reese et al. 2023). Similarly, breeding-season parkwide surveys within MEVE conducted by SCPN from 2022 to 2024 detected 117 species using the park as breeding or stopover habitat (H. Jones, unpublished data).

Three raptor species of conservation interest in MEVE include the peregrine falcon (*Falco peregrinus*), golden eagle (*Aquila chrysaetos*), and Mexican spotted owl (*Strix occidentalis lucida*). The peregrine falcon typically lives in open landscapes near cliffs and canyons, where it typically “dive bombs” its prey (usually other, smaller birds) from above (Stevens et al. 2009). The peregrine falcon population in Colorado was reduced to five breeding pairs in the 1970s, leading to rigorous translocation efforts. By the early 2000s, its population had increased approximately 10-fold (Craig and Enderson 2004). Golden eagles in the West occupy wide open spaces, nesting on cliffs or trees (Kochert et al. 2002). Their primary prey are rabbits, hares, ground squirrels, and prairie dogs (Kochert et al. 2002). This globally distributed species is of great cultural importance to many Native American tribes and is protected under several federal laws, including the 1940 Bald and Golden Eagle Protection Act. Mexican spotted owls were listed as a federally threatened species by the U.S. Fish and Wildlife Service (USFWS) in 1993 (Mexican Spotted Owl Recovery Team 2012) due to habitat modification and climate-related changes to the forest vegetation with which they and their prey species are associated. Spotted owls nest in deep canyons, and while their primary prey includes small terrestrial mammals, they also hunt birds, bats, arthropods, and reptiles (USFWS 2012). MEVE is on the northern edge of the species’ range in the American Southwest.

Hummingbirds (Trochilidae) are another bird family of conservation significance in the Park. MEVE’s four most common hummingbird species include two breeding species, the black-chinned (*Archilochus alexandri*) and the broad-tailed hummingbird (*Selasphorus platycercus*), and two migrant species, the calliope (*Selasphorus calliope*) and the rufous hummingbird (*Selasphorus rufus*). These are nectar-feeding species that have an affinity for Indian paintbrush (*Castilleja coccinea*) and several other flowering species. Hummingbirds hold importance in numerous Indigenous cultures across the Northern and Southern hemispheres, including on the Colorado Plateau (Eckert and Clark 2009; Ferg 2007).



Mexican spotted owl.
NPS / K. BARNETT

Raptors Gap Analysis Summary

Golden Eagle. MEVE has a comprehensive database of golden eagle sightings in the Park dating from the mid-1920s to 2001. This long-standing eagle dataset included results both from periodic (typically annually in winter) eagle surveys, as well as incidental sightings reported throughout the year. Though these data lacked available metadata, or descriptions of how the data were obtained, and as such must be interpreted with caution, they provide an estimate of changes in eagle detections through time. Figure 6 depicts a marked increase in golden eagle detections in the Park beginning around 1980, peaking in the late 1980s, and then dropping to around pre-1980 levels in the early 2000s before detections stopped being recorded in the database. We note that the increase in eagle sightings in the 1980s and 1990s was confirmed to be a likely artifact of increased survey effort during those two decades (personal communication, Nathan Brown and Paul Morey, Mesa Verde NP, January 2025).

While input into the MEVE golden eagle database ended in 2001, additional information from periodic eagle surveys was available from the MEVE Natural Resource or Wildlife Reports for 2002 through 2022. We considered data from these reports separately from the golden eagle database given that only annual survey results (and not opportunistic sightings) were included in the reports. Based on these surveys, 10–14 golden eagles were reported in MEVE between 2002 and 2007 (though no survey was conducted in 2004). In the 2008 survey, seven eagles were reported, and in 2009 there were five. From 2010 through 2023, the winter golden eagle surveys were conducted less often, occurring in seven of those 13 years. During that period, between 3 and 5 eagles were reported per survey, including at least one fledgling recorded from at least one nest from 2018 to 2023, except for 2020 (personal communication, Nathan Brown, MEVE NP Biologist & Wildlife Program Manager, January 2025).

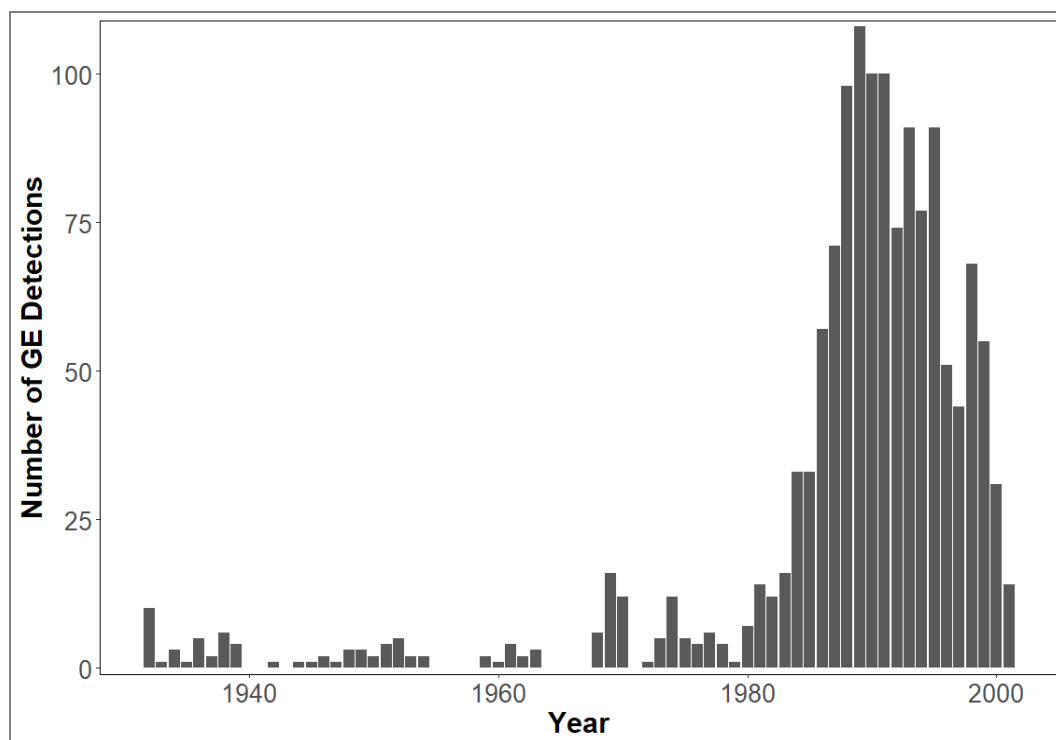


Figure 6. Golden eagle (GE) detections per year from 1932 to 2001 based on MEVE records. Created using NPS data.

NPS / CONSERVATION SCIENCE PARTNERS

USFWS has developed a model predicting the spatial variation in density of golden eagle nest sites across the western United States (Dunk et al. 2019). The model was created utilizing >6,500 known nest site locations and modeled for twelve distinct regions. We used the southwestern plateau regional model (Figure 7a) and assessed two model products to understand MEVE’s predicted golden eagle density. Relative Nest Site Density (RND) (Figure 7b) is the raw continuous dataset representing the predicted nest density within each modeled region. Area-adjusted Frequencies (AAF) (Figure 7c) is the extent to which the number of golden eagle sites differs from a random distribution of nests within the modeled region and allows for direct comparisons of locations. Based on the results of the RND model within MEVE, there are many locations of high predicted relative nest density (Figure 7b). These locations are primarily found in the northwest and southwest sections of the Park on cliff tops, which is a typical nesting habitat for golden eagles (Kochert et al. 2002). Results from the AAF model are similar, as locations determined to be “very high density” are also located in typical nesting habitat within the southwest region of the Park (Figure 7c).

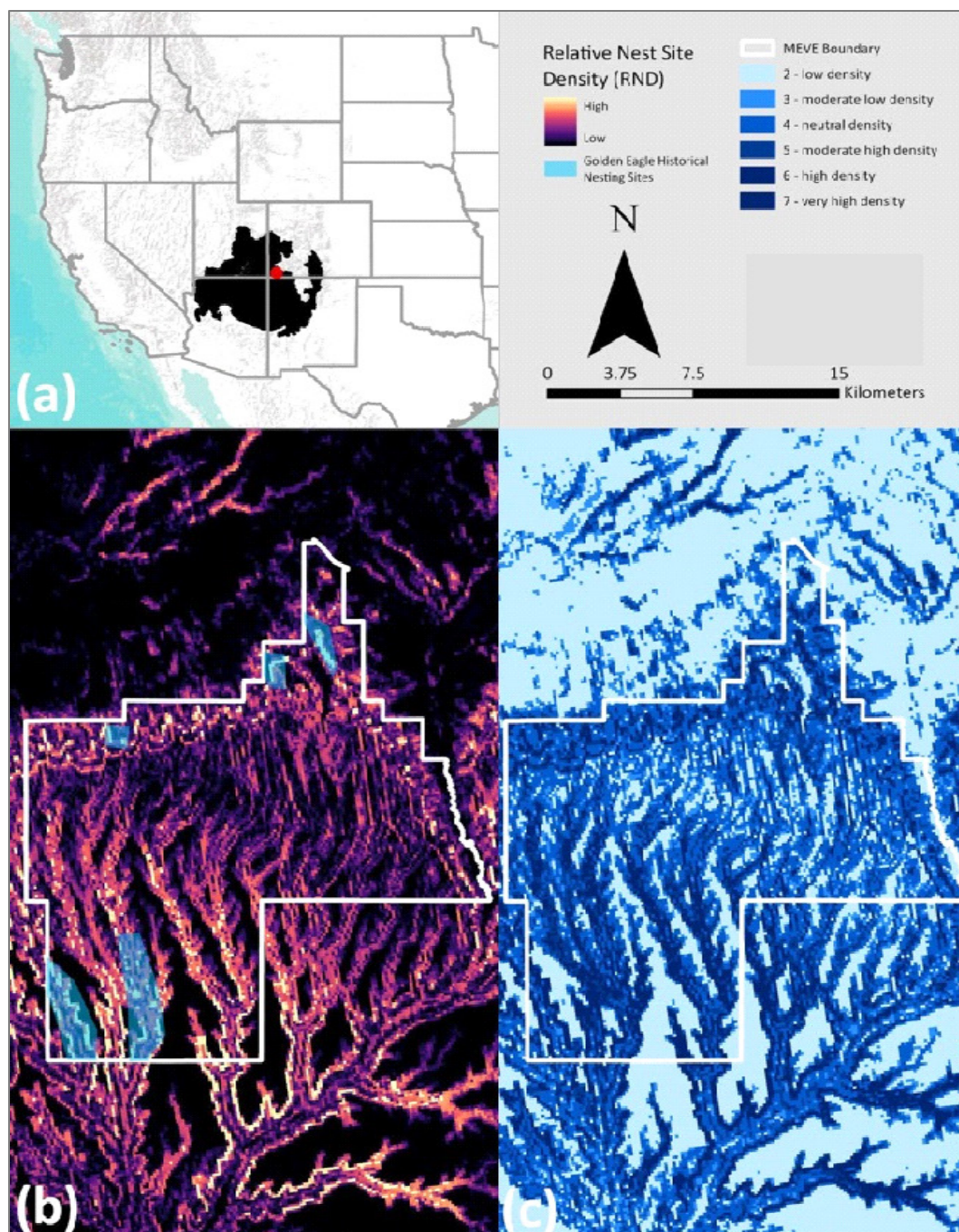


Figure 7. Predicted relative density of golden eagle nest sites, based on spatial layers developed by USFWS. a) Distribution of the Southwestern Plateau regional model developed by Dunk et al. (2019). MEVE is represented as the red dot. b) USFWS Relative Nest Site Density (RND) model product mapped across the boundaries of MEVE. General locations of historical nesting sites are highlighted as blue translucent polygons (see text for details). c) USFWS Area-adjusted Frequencies (AAF) model product mapped across the boundaries of MEVE. Created using data from DUNK et al. (2019).

NPS / CONSERVATION SCIENCE PARTNERS

We collected information on documented golden eagle nesting sites utilizing historical records from Colorado Parks and Wildlife (CPW), and first-hand accounts from Marilyn Colyer, a previous MEVE Natural Resources Biological Technician. Due to the lack of distinct spatial coordinates of these nesting sites, we created a layer generalizing their locations (documented in Figure 7b as translucent polygons based in Navajo Canyon cliffs, Rock Canyon, Point Lookout cliffs, Point Lookout, Knife Edge Trail, and the North Escarpment). After overlaying the RND model within MEVE and the nesting site polygons, we found considerable overlap between historical records and areas of high predicted nest density.

To assess MEVE's predicted golden eagle nest site density in relation to the entire model distribution, we calculated the total area of the southwest plateau and of MEVE that is considered "high density" or "very high density" according to the AAF model. MEVE has a disproportionately large percentage of high/very high-density habitat (17% of the Park) compared to the whole southwestern plateau (4% of the total model area). Therefore, MEVE is important for the golden eagle population in this region of the West due to its ability to provide accessible nesting habitat.

Lastly, we explored local golden eagle trends in the area, as predicted by a range-wide model developed with the eBird citizen science database compiled and maintained by the Cornell Lab of Ornithology (eBird, 2024). The model uses a grid of 27km × 27km regions. Two of these grid cells overlap the northern and southern sections of the park (eBird 2024). For the period 2012–2022, e-Bird reported that the trend for golden eagles at the southeast Park border cell was –16% (CIs –1.4%, –24%), and the reported trend for the cell on the Park's northern boundary was –19% (CIs –0.66%, –30%) (Fink et al. 2023).

Peregrine Falcon

The status and condition of the peregrine falcon population that utilizes MEVE and surrounding territories is currently unknown. However, there is evidence that the Park has supported, and might continue to support, 1–3 pairs of peregrines at any given time. Park staff began surveying for peregrines in the 1970s. Starting in 1982 the annual surveys were conducted by the Peregrine Fund, which continued until 2000 when the annual MEVE peregrine surveys were taken over by CPW (Colyer and SanMiguel 2001). As summarized in the 1982–1999 Peregrine Fund survey reports, the Knife Edge eyrie was consistently surveyed and found to have nesting pairs and/or fledglings during this time (Colyer and SanMiguel 2001). There are records of the surveys that were conducted by MEVE staff from 1997 through 2008, summarized in the MEVE annual Resource Management reports. During this time period surveyors would visit anywhere from one to 10 known or suspected eyries, typically finding paired adults at about two-thirds of the eyries surveyed (Mesa Verde National Park 1997, 1998, 1999, 2000, 2001, 2002, 2003, 2004, 2005, 2006, 2007, 2008). During those years, MEVE reported anywhere from 3 to 9 total fledglings a year from two to five eyries.

Funding for the CPW peregrine falcon surveys was cut in 2000, shortly after the species was removed from the Endangered Species list. At this point MEVE took over the peregrine surveys, but limited resources often only allowed a check of the dependable Knife Edge Eyrie, especially from 2009 through 2020. From 2010 through 2014 it is not clear whether Knife Edge was visited annually

by Park staff, but if/when it was visited no peregrines were detected. From 2015 through 2017 there was an increased effort by MEVE to survey Knife Edge; peregrines were documented nesting there during those years, with fledglings documented in 2015. 2020 was the last year peregrines have been seen nesting at Knife's edge (Brown 2021). Notably, prairie falcons were seen at the Knife Edge cliffs in 2021 and 2022 (personal communication, Nathan Brown, MEVE NP Biologist & Wildlife Program Manager, March 2024), and detected within the park by SCPN survey crews in 2022 and 2024.

Mexican Spotted Owl

While there are records of Mexican spotted owl (MSO) detections in MEVE dating back to the 1940s, deliberate nocturnal calling surveys for the species began in the early 1980s. As such, there is a comprehensive database of all Mexican spotted owl detections in the Park dating from the early 1980s to the early 2000s. The number of detections generally increased from the early 1980s to the early 1990s and then began decreasing (Figure 8). Unfortunately, the surveys, though continuing to the present, stopped eliciting return calls from owls in the early 2000s. According to MEVE records, the last confirmed Mexican spotted owl was documented in the Park in 2003. The last reported detection was in 2009 when a Park biologist reported hearing a male and female spotted owl, though follow up surveys could not confirm their presence.

Based on these survey results, it appears that Mexican spotted owls have been extirpated from MEVE sometime within the last two decades. Figure 9 depicts spotted owl detection sites in the Park over time and shows an apparent preference for the northeast and southern boundaries of MEVE.

MEVE is on the northern periphery of the occupied range of the species. The population of MSO in the Park has always been small compared to populations elsewhere on the Colorado Plateau, such as the Mogollon Rim in east-central Arizona or the Gila mountains in New Mexico. It is thought that only a few breeding pairs of owls occupied the park at any time (personal communication, Nathan Brown, MEVE NP Biologist & Wildlife Program Manager, September 2023). With average dispersal distances of 25–30 km (Willey and VanRiper III 2000) and only occasionally more than 75 km (Ganey et al. 1998), and with the closest occupied MSO territories either just under 100 km away on the Navajo nation or over 100 km away in the Carson National Forest in northern New Mexico or in the Abajo Mountains/Bears Ears region of southeast Utah, it could be argued that a sustainable population of owls in MEVE was always tenuous. In addition, the small population that was supported in the Park was likely at least somewhat reliant on the comparatively small patches of Douglas fir, and over time those patches have diminished (personal communication, Nathan Brown, MEVE NP Biologist & Wildlife Program Manager, September 2023) due to drought-driven wildfires and bark beetle outbreak.

While MSO has apparently been extirpated from MEVE, the general distribution of the meta-population of MSO on the Colorado Plateau Ecological Management unit has not changed since listing range-wide in the U.S. (U.S. Fish and Wildlife Service 2023). However, in their most recent status review of the species, the USFWS concluded that the owl would remain a threatened species due to climate change and extended drought that affect habitat suitability and prey availability, as

well as the threat of habitat alteration and destruction caused by wildfire that continues to increase in pace and scale because of climate change (U.S. Fish and Wildlife Service 2023).

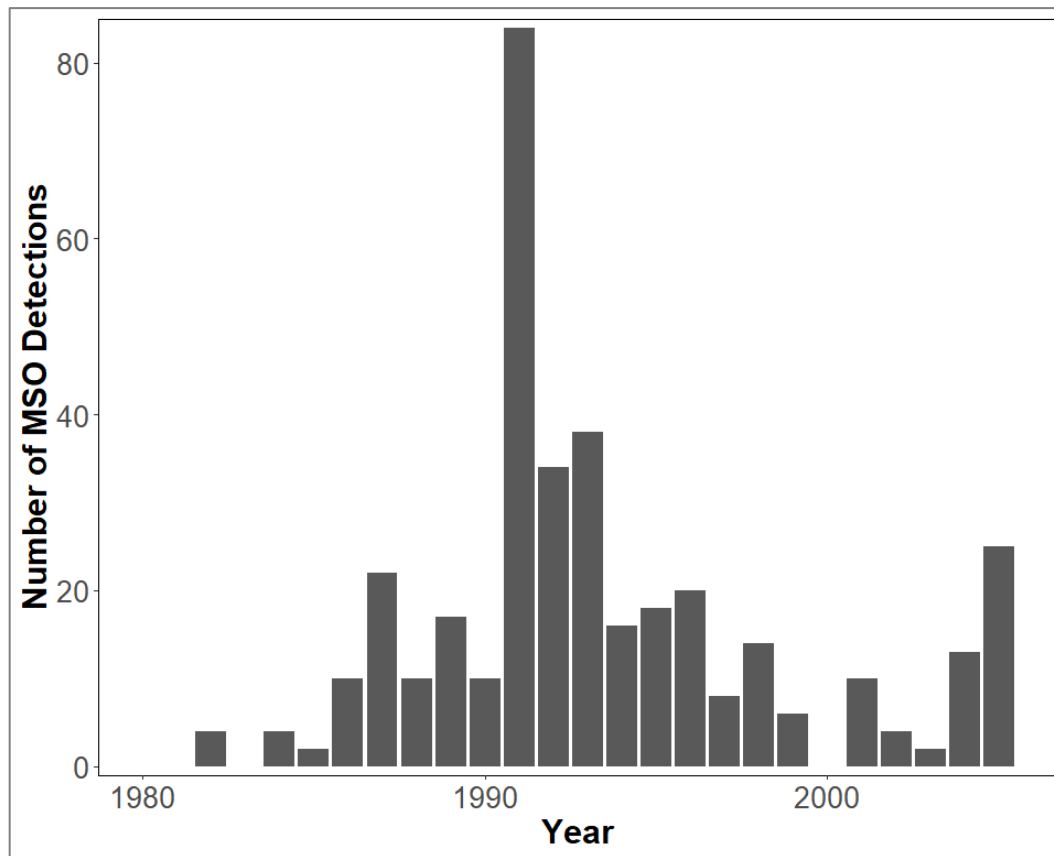


Figure 8. Mexican Spotted Owl (MSO) detections per year from 1980 to 2005 based on MEVE records. Created using NPS data.

NPS / CONSERVATION SCIENCE PARTNERS

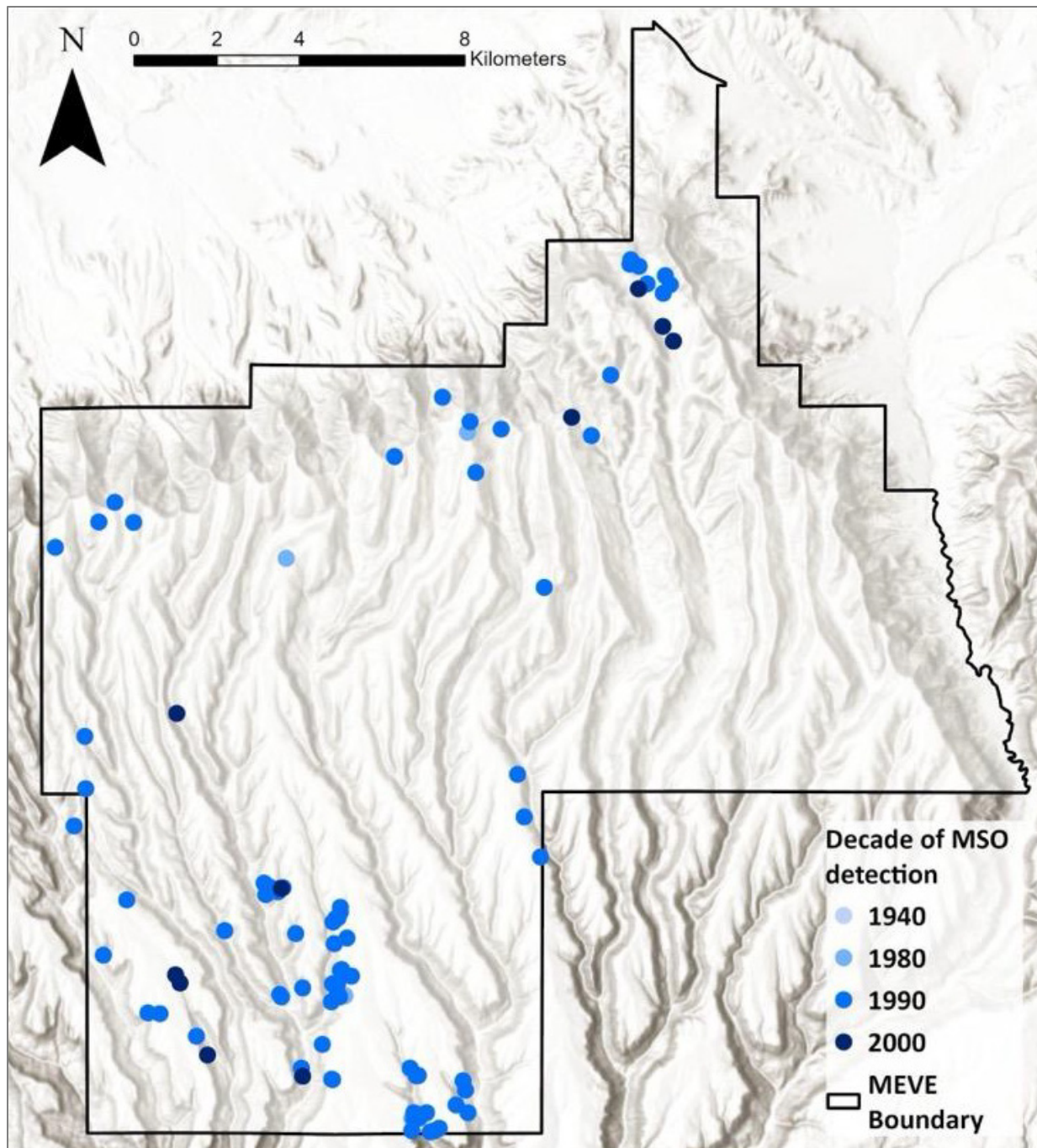


Figure 9. Sites where Mexican spotted owls have been detected in MEVE, color coded for the decade of last detection. Data from NPS: MEVE records. Created using NPS data.

NPS / CONSERVATION SCIENCE PARTNERS

Passerines Condition Assessment Summary

“Snapshot” of Current, Common Passerine Species Densities

We modeled breeding season density for 14 common bird species associated with MEVE’s PJ woodland and Gambel oak shrubland habitat using a Bayesian hierarchical model following

Amundson et al. (2014) (Figure 10). Details for analytical and statistical methods utilized for this and all subsequent passerine analyses can be found in Appendix B. We utilized point count data from 2022 and 2023 collected by NPS’s Southern Colorado Plateau Network (SCPN) (Ploughe & Jones, 2024) for this analysis. We found an overall breeding season density of ~ 0.5 birds per ha^{-1} for species included in our model (0.45 ± 0.54 ; mean $\pm$ SD of mean values). Spotted towhee (*Pipilo maculatus*) had the highest density across all species (2.17 ± 0.09 birds per ha^{-1}), followed by blue-gray gnatcatcher (*Polioptila caerulea*; 0.75 ± 0.05 birds per ha^{-1}) and black-throated gray warbler (*Setophaga nigrescens*; 0.72 ± 0.06 birds per ha^{-1}). Gray vireo (*Vireo vicinior*), which is a Partners in Flight bird of conservation concern for the state of Colorado, had the lowest density (0.08 ± 0.02 birds per ha^{-1}) across all species (Crafford et al., In preparation).

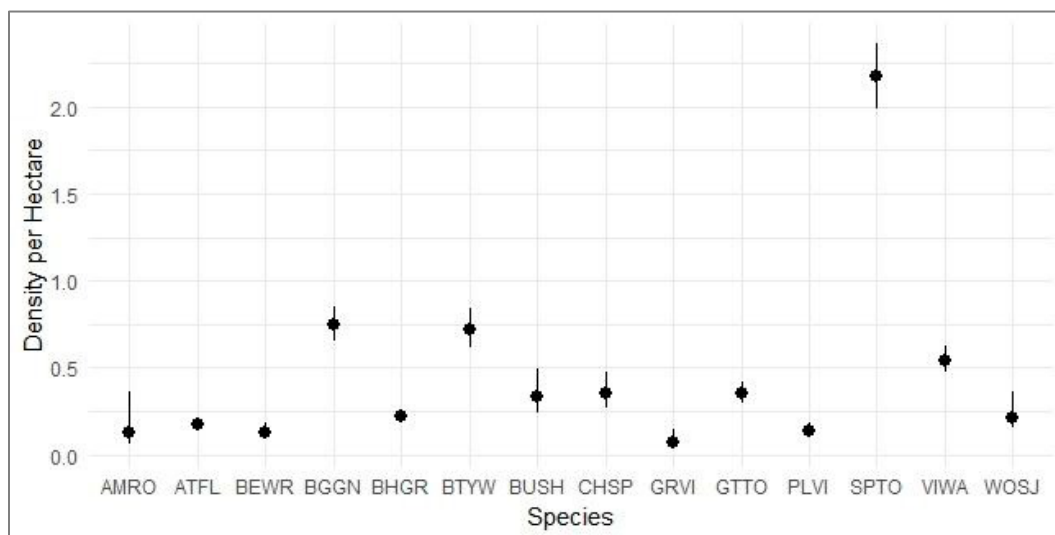


Figure 10. Density per hectare of most common passerine species in MEVE. Error bars indicate the 95% Bayesian Credible Interval for each species. Appendix C includes the common and scientific names associated with the 4-letter bird codes. Created using data from SCPN (Ploughe & Jones 2024).

NPS / CONSERVATION SCIENCE PARTNERS

Jones et al. (2024) recently reported density trends from 2008 to 2018 for MEVE bird species. Of the 21 species modeled, seven (broad tailed hummingbird, gray flycatcher [*Empidonax wrightii*], plumbeous vireo [*Vireo plumbeus*], Bewick’s wren [*Thryomanes bewickii*], house finch [*Haemorrhous mexicanus*], lesser goldfinch [*Spinus psaltria*], and chipping sparrow [*Spizella passerina*] showed significant population growth rates over the course of the decade. Blue-gray gnatcatcher and gray vireo population densities remained relatively stable from 2008 to 2018 (Jones et al. 2024). Finally, juniper titmouse (*Baeolophus ridgwayi*) and spotted towhee showed statistically significant decreases in population density during that time (Jones et al. 2024). This result is interesting given that spotted towhee had the highest density across all species based on the above model results, which incorporated 2022 and 2023 survey years.

“Snapshot” of Avian Species Trends Over Time

Utilizing the Integrated Monitoring in Bird Conservation Regions’ (IMBCR) estimation of passerine trends derived from their 2008–2022 MEVE point count dataset (Pavlacky Jr et al. 2017), we summarized these estimated population trends in Appendix D. IMBCR calculated population trends for 76 species (where a value of 1 indicates the population is stable, < 1 indicates the population is declining, and > 1 is an increasing population, over the 15-year period). After removing species that had fewer than five detections from 2008 to 2022, we found that 11 of 53 species had overall trends of 1.10 or more over the time period, indicating annual rates of increase of 10% or more per year (Appendix D). Bushtits (*Psaltirparus minimus*), with a total of 23 detections over the study period, exhibited the greatest rate of increase, with a median population trend of 1.21 from 2008 to 2022, or a 21.1% increase per year. Twelve of 53 species had overall population trends of 0.90 or less over the time period, indicating annual rates of decrease of 10% or more per year. Northern mockingbirds (*Mimus polyglottos*), with a total of 25 detections over the study period, exhibited the greatest rate of population decrease, with a median trend of 0.68 from 2008 to 2022, or a 32.3% decrease per year (Appendix D). However, since the northern mockingbird is currently experiencing range-wide declines (Sauer et al. 2019), this is perhaps not surprising.

Passerine Species Richness

Utilizing the IMBCR 2008–2022 point count dataset (Pavlacky Jr et al. 2017) and the SCPN 2007–2018 point count dataset (Ploughe and Jones 2024), we summarized species richness (number of total species detected) in MEVE from 2007 to 2022 (Figure 11). The IMBCR surveyed 80 sites in various habitat types (principally PJ or mountain shrub/Gambel oak) in the months of May, June or July. The SCPN surveyed 86 sites only in undisturbed PJ woodlands in the months of May, June or July, with two replicates per year. Species richness varied from 29 (2018) to 59 total species (2009) recorded in the IMBCR surveys, and 49 (2018) to 62 total species (2009) recorded in the SCPN surveys (Figure 11). Species richness varied across the years and was highest in 2009 and lowest in 2018 in both datasets. In light of these findings, it is important to note that SCPN and IMBCR sampling protocols capture a subset of the full breadth of habitats in the Park. Thus, species richness numbers may underestimate richness for the Park as a whole due to lack of sampling in all habitats, and across an elevational gradient.

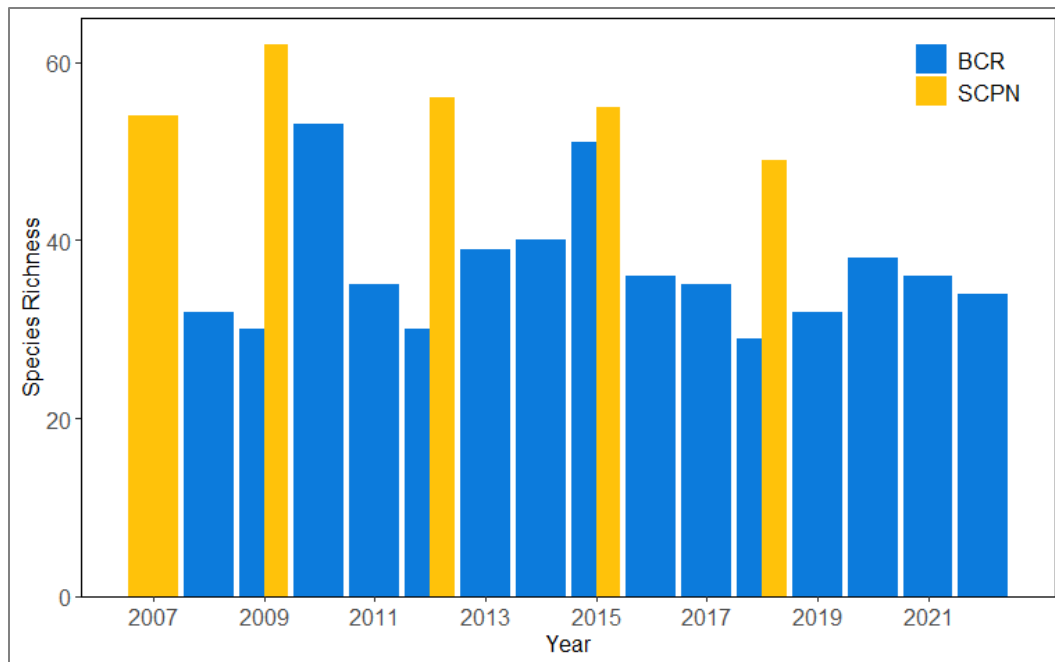


Figure 11. Species richness (number of total species) per year from 2007 to 2022 based on data from the IMBCR and the SCPN. The BCR completed 80 surveys each year from 2008 to 2022, and the SCPN completed 86 surveys every 2–3 years from 2007 to 2018. Created using data from the IMBCR (Pavlacky Jr et al. 2017) and the SCPN (Ploughe and Jones 2024).

NPS / CONSERVATION SCIENCE PARTNERS

Comparing Trends in IMBCR Data for MEVE to Larger Region

We also acquired IMBCR (Pavlacky Jr et al. 2017) population estimates (including species occupancy, density, and population growth rate) for the suite of avian species that exist in the Colorado portion of the Southern Rockies/Colorado Plateau Region (i.e., CO-BCR-16) for context with estimates in MEVE. Figure 12 shows that the majority (30) of 40 species that are found in both the Park and CO-BCR-16 have higher occupancy rates in MEVE than those averaged across CO-BCR-16. In fact, in 22 cases, the average occupancy rates in the Park are double or more the rates in CO-BCR-16.

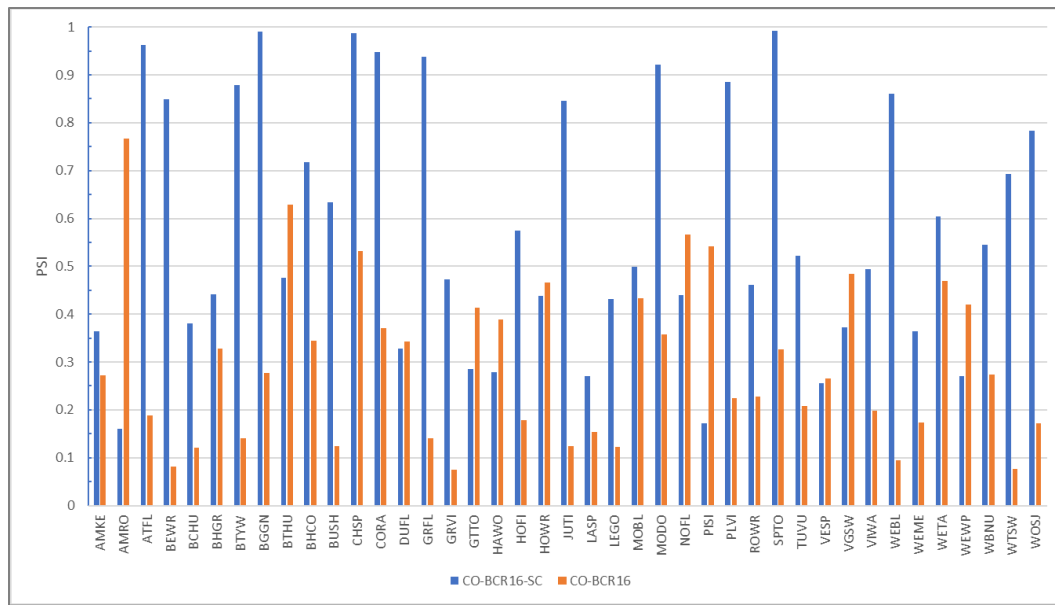


Figure 12. Average occupancy rates (from 2008 to 2022) of 40 common bird species in MEVE (CO-BCR16-SC, shown in blue) compared to average occupancy rates for those species for the same time period in the broader Colorado-portion of the Southern Rockies/Colorado Plateau region (CO-BCR 16, shown in orange). Species four-letter codes and associated common names in Appendix C. Created using data from the IMBCR (Pavlacky Jr et al. 2017).

NPS / CONSERVATION SCIENCE PARTNERS

Passerine Response to Fire & Habitat Attributes

We quantified the responses of passerine bird communities to fire-induced changes in habitat structure and composition. Many areas in MEVE previously comprising PJ woodland are now dominated by Gambel oak (*Quercus gambelii*) shrubland, and while this successional trajectory is not unexpected, recent studies have found a lack of PJ regeneration in these areas (Phillips et al. 2023). Further, in many other locations, burned former PJ woodlands have transitioned to grassland (Phillips et al. 2023), which raises questions about how PJ-associated bird species are responding to these habitat-level shifts. As such, we were interested in determining how breeding season densities of common species were affected by habitat structure, vegetation composition, whether or not a habitat was burned, and post-fire succession (represented as the variable “time-since-fire”).

We utilized 2022–2023 SCPN point count data from burned and unburned parts of MEVE to determine bird community composition in these areas (Ploughe and Jones 2024). We extracted two metrics of vegetation structure, canopy cover and vegetation height, from aerial, discrete return LiDAR (Light Detection and Ranging) point clouds downloaded from the Colorado Water Conservation Board website (Colorado Water Conservation Board 2024). We applied a custom function (Crafford et al. In preparation) based on vegetation height thresholds using a filtered LiDAR point cloud to extract our vegetation metrics of interest. We also quantified the presence of deciduous vegetation (i.e., Gambel oak, antelope bitterbrush [*Purshia tridentata*]) using cloud-free satellite imagery mosaics from the Landsat 8 Operational Land Imager (OLI), using Collection 2 surface reflectance data (Vermote et al. 2016). Fire history and perimeter data were gathered from the

National Interagency Fire Center's online repository (National Interagency Fire Center 2024). All remotely sensed data were ground-verified using coarse vegetation classifications taken in the field at each survey point. Further details on the methodology can be found in Appendix B.1.

We found that four MEVE bird species (black-headed grosbeak [*Pheucticus melanocephalus*], black-throated gray warbler, Virginia's warbler [*Leiothlypis virginiae*], and spotted towhee) showed statistically supported responses to higher percent canopy cover; black-throated gray warbler increased in density with greater cover, while the three others decreased (Figure 13). Deciduousness, which here is used as a proxy for presence of deciduous shrubs such as Gambel oak (and absence of coniferous vegetation like PJ), affected the densities of nine (64%) MEVE birds; American robin (*Turdus turdidae*), blue-gray gnatcatcher, black-headed grosbeak, Virginia's warbler, plumbeous vireo, spotted towhee, and green-tailed towhee (*Pipilo chlorurus*) all increased in density with deciduousness, while black-throated gray warbler and Bewick's wren decreased. Finally, black-headed grosbeak, bushtit (*Psaltirparus minimus*), ash-throated flycatcher (*Myiarchus cinerascens*), and Virginia's warbler increased in density with the presence of tall trees (vegetation height > the 95th percentile) (Figure 13).

Five (36%) MEVE birds (American robin, blue-gray gnatcatcher, black-throated gray warbler, chipping sparrow, and green-tailed towhee) showed statistically supported decreases in density in burned areas. Interestingly though, six (43%) MEVE bird species increased in density with greater time-since-fire, while one decreased. We saw increases in blue-gray gnatcatcher, black-throated gray warbler, bushtit, gray vireo, Woodhouse's scrub jay (*Aphelocoma woodhouseii*), and spotted towhee; American Robin decreased (Figure 13).

Based on our findings, it is clear that burned habitat has a negative effect on the densities of some common MEVE bird species, though many MEVE birds also increased with time-since-fire. This suggests that these birds may not be entirely lost from the landscape if PJ does not return, since the post-fire landscape to which these species are responding positively is markedly different from PJ woodland or shrubland. This is also supported by the fact that many MEVE birds increased alongside deciduousness. However, there are structural aspects to the landscape, such as tall trees, that are important for a handful of MEVE birds, which indicates that some species' persistence in post-fire landscapes hinges on successional outcomes; in other words, a complete shift from woodland to native or non-native dominated grassland, as many burned parts of MEVE have experienced, would be detrimental for these woodland species. Furthermore, species that decreased with burned habitat but did not show supported increases in density with time-since-fire (e.g., green-tailed towhee and chipping sparrow), or species that had a negative relationship to deciduousness (Bewick's wren, black-throated gray warbler) may be most vulnerable if PJ does not regenerate following fire.

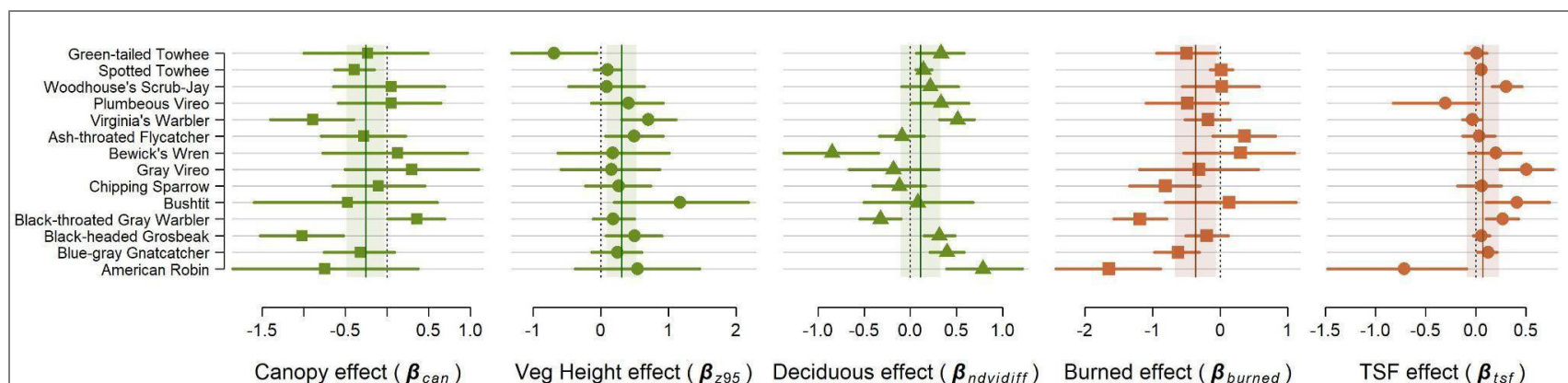


Figure 13. Effects of vegetation structure and composition and fire coefficients on breeding season abundances for 14 common MEVE passerines. Mean and 90% Bayesian credible intervals (BCI) are shown for each species. Overall mean and standard error of mean species level effect sizes are represented by a solid vertical line and the shaded area in each plot, respectively. If the BCI for a species does not cross the dashed vertical line at zero, that species showed a statistically supported response to that covariate. TSF effect shows the effect of time-since-fire. Created using data from the SCPN (Ploughe and Jones 2024).

NPS / CONSERVATION SCIENCE PARTNERS

Hummingbirds Condition Assessment Summary

A 2007–2022 banding dataset supplied by the Hummingbird Monitoring Network (HMN) was used for our hummingbird analyses (Wethington 2023). This dataset was collected through a standard protocol that is used at all HMN monitoring and banding sites throughout the West. There is one HMN site within MEVE, located in the Morefield Campground. At this site, five hummingbird feeders are maintained during the hummingbird occupancy season. Each year from late May to early September, hummingbirds are captured (using Hall traps that cover the feeders) and banded one morning every other week. Trapping and banding begin within one half-hour of sunrise and continue for five hours.

For the hummingbird analyses we focused on the two MEVE breeding species, black-chinned (*Archilochus alexandri*) and broad-tailed hummingbirds (*Selasphorus platycercus*). We did not include non-breeding hummingbird species in this analysis because the Park is small enough that declines in migratory species may have nothing to do with Park conditions. We analyzed the trends in number of captures of these two species banded in MEVE from 2007 through 2022 (Figure 14) utilizing a Generalized Least Squares (GLS) regression model. We utilized GLS as it accounts for non-independence between observations, as found in this dataset. GLS models were applied separately to each species and all models accounted for autocorrelation between yearly observations. Assumptions were tested and residuals were normally distributed (Shapiro test; p -value > 0.05) and no significant residual autocorrelation was detected (Ljung-Box test p -value > 0.05). The GLS model results for both species resulted in non-significant slopes (black-chinned: $\beta_1 = 0.5640$, p -value = 0.9524 ; broad-tailed: $\beta_1 = -1.3097$, p -value = 0.0968). These results suggest that both species were stable over this period of time.

We also visually compared the 2007–2022 black-chinned and broad-tailed hummingbird data provided by the HMN to the point count data for the same two hummingbird species provided by the SCPN (years 2007–2018) (Ploughe & Jones, 2024) and the IMBCR program (years 2008–2022) (Pavlacky Jr et al. 2017). We visually examined the point count data for these two species for these datasets and, similar to the HMN banding data, detected no trends in relative abundance over the timeframe in question.

The current condition status of indicators of 1) population trends of 53 birds tracked by IMBCR; 2) occupancy rates of MEVE species compared to regional (CO-BCR-16) occupancy rates; 3) passerine responses to fire; 4) passerine response to time-since-fire; and 5) trends of the two breeding hummingbird species were assessed using the condition rating thresholds developed in Appendix A.1. Table 2 below illustrates the outcome of this exercise, based on the data summarizations and analyses for birds presented above.

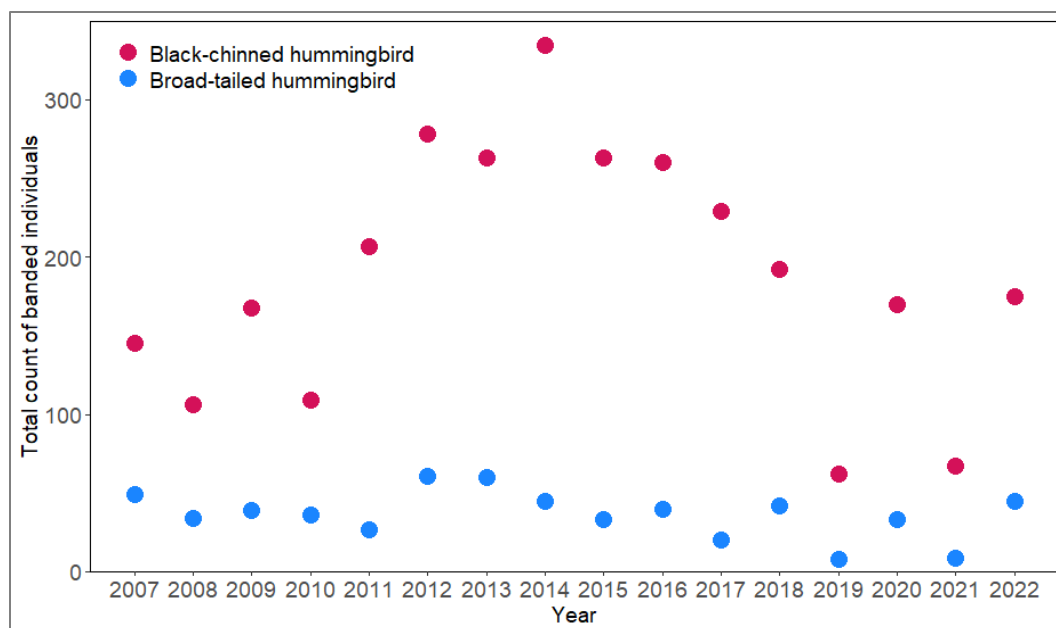


Figure 14. Annual count data of broad-tailed hummingbirds and black-chinned hummingbirds banded in MEVE from 2007 to 2022. GLS model results indicated stable populations over the study period for both species. Created using data from the HMN (Wethington 2023).

NPS / CONSERVATION SCIENCE PARTNERS

Table 2. MEVE passerines and hummingbirds condition ratings summary. Condition ratings below are based on the Condition Rating Table in Appendix A.

Indicator	Condition	Trend ^A	Condition Evaluation
Population growth rates of 53 MEVE species (2008–2022)	Fair	N/A	The condition is fair because there are roughly equal numbers of species with positive population growth rates of 10%/yr or more, versus negative trends of 10%/yr or more, over the study period.
Occupancy rates of birds in MEVE compared to BCR-16	Good	N/A	The condition is good because the majority of species have higher occupancy rates in MEVE than in the larger region.
Passerine response to fire	Fair/Poor	N/A	The condition is fair/poor because 36% of 14 modelled MEVE passerines assessed decreased in density in response to burned habitat.
Passerine response to time-since-fire	Fair	N/A	The condition is fair because 43% of 14 modelled MEVE passerines assessed increased in density as time-since-fire increased.
Hummingbird trends	Good/Fair	N/A	The condition is good/fair because trends of both BTHL and BCHU are stable according to GLS regression models, despite some fluctuations over time.

^A The nature of the data summarized and analysis performed for birds did not lend itself to an assessment of trend for these indicators.

3.2. Wildlife—Deer and Elk

Two species of ungulates are found within MEVE: elk (*Cervus elaphus*) and mule deer (*Odocoileus hemionus*). Both species have ranges that include northern and eastern portions of MEVE. Elk may migrate between alpine habitats during the summer/early fall, and lower elevation piñon-juniper (PJ) and sagebrush habitats during the winter, depending on weather conditions. Within the United States, the elk population plummeted to its lowest recorded point, approximately 40,000 individuals, in the early 1900s because of unregulated market hunting (Colorado Parks and Wildlife n.d.). Since that time, Colorado, like other states, has imposed hunting and management regulations and has conducted extensive translocation efforts. In 2022, Colorado had 280,000 elk, the largest population of elk in the world (Colorado Parks and Wildlife n.d.).

Mule deer are found within a diversity of habitats in the western United States including shrublands and a variety of forest types. Despite their habitat generalism, mule deer populations have decreased in Colorado for decades. By 2012, CPW estimated the state's population to be 408,000 mule deer, 22–29% below their state management goal (Colorado Parks and Wildlife 2017).



Bull Elk.

NPS

Deer and Elk Gap Analysis Summary

CPW Deer and Elk Population Estimates

For the following gap analysis for deer and elk, we relied on annual population estimates generated by CPW. To achieve these estimates, CPW utilizes spreadsheet models that incorporate: (1) sex ratio and age ratio data gathered for each Data Analysis Unit (DAU) annually through winter classification flights, (2) hunter harvest data gathered through annual surveys in each DAU, and (3) survival rates

for calves/fawns and adults that are monitored annually at several locations within the state (personal communication, Brad Weinmeister, Wildlife Biologist, Colorado Parks and Wildlife, September 2024). Each year, when these data are combined, CPW recalculates the current population estimate for both deer and elk for each DAU.

Elk Gap Analysis

CPW provided data on the post-hunt population estimates for elk (Colorado Parks and Wildlife 2024c) in Colorado's DAU E(Elk) 24, which includes Game Management Units (GMUs) 70, 71, 711, 72 and 73, for the herd that overlaps MEVE. There has been a slight upward trend in these herd population estimates for elk in this DAU over the past two decades. Because precipitation variation can affect population levels of ungulates through deep snow that can increase winter mortality, or strong spring green-up that can affect fawn survival rates, we compared elk population estimates over time to changes in annual precipitation (both rain and snow) for the region. Elk numbers were slightly higher in 2021 compared to 2004 and showed no clear response to precipitation during that same period.

Deer Gap Analysis

CPW provided data on the post-hunt population estimates for deer (Colorado Parks and Wildlife 2024b) in Colorado's DAU D(Deer) 29, which includes Game Management Units (GMUs) 72 and 73, for the herd that overlaps MEVE. There has been a slight upward trend in these herd population estimates for deer in this DAU over the past two decades (Figure 15). We compared deer population estimates over time to changes in annual precipitation (both rain and snow) for the region (Figure 15). Deer numbers were slightly higher in 2022 compared to 2004 and showed no clear response to precipitation during that same period.

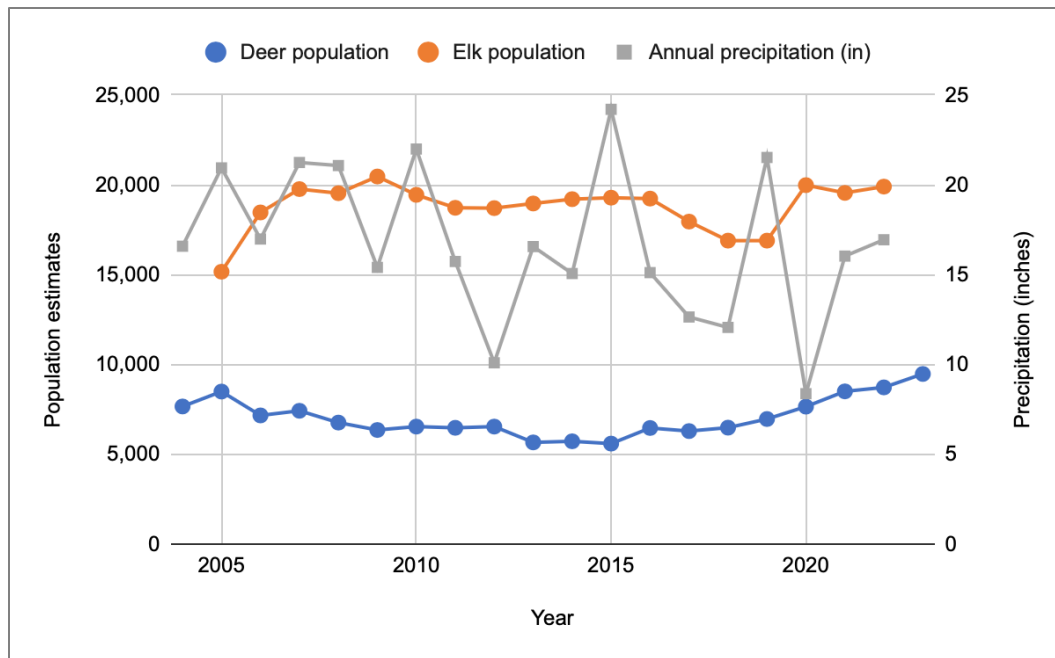


Figure 15. Annual estimation of deer and elk herd sizes (for the herd in the Game Management unit that includes MEVE) from 2004 to 2022 compared to annual precipitation (in). Created using deer & elk data from the CPW and precipitation data from the National Weather Service (N.D.).

NPS / CONSERVATION SCIENCE PARTNERS

Ungulate Habitat Overlay

We overlaid regional elk and mule deer migration corridors and winter and summer ranges within CPW DAU E24 and DAU D29 (Colorado Parks and Wildlife 2024a). MEVE contains a small portion of winter range for elk, both summer and winter range for mule deer, and connects to the portions of those ranges that extend beyond the park via both winter range habitat and ungulate migration corridors identified by CPW (Figure 16).

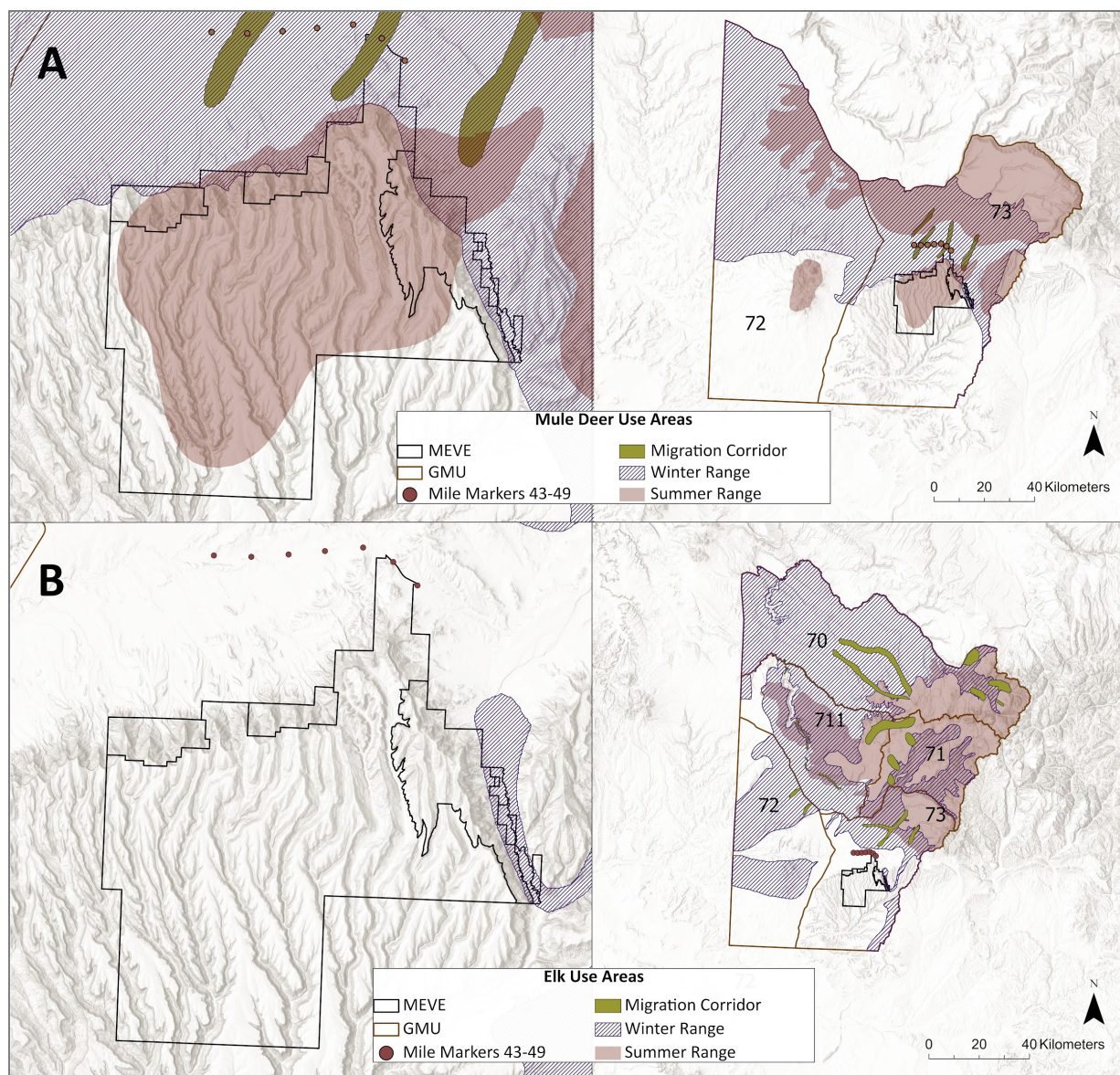


Figure 16. Overlay of elk and mule deer migration corridors and winter and summer ranges within CPW Data Analysis Unit (DAU) E24, which consists of Game Management Units 70, 71, 711, 72, and 73 and DAU D29. Panel A, right: Mule Deer use of Mesa Verde; panel A, left: Mule Deer use across DAU D29. Panel B, right: Elk use of Mesa Verde; panel B, left: Elk use across DAU E24. Data from the CPW Hunting Atlas (Colorado Parks and Wildlife 2024a). Highway mile markers in B denote high priority wildlife crossing areas (see text for details). Created using data from the CPW.

NPS / CONSERVATION SCIENCE PARTNERS

Because of this important connection between MEVE and adjacent elk and deer habitat, we highlight mile markers 43–49 of Highway 160 on the north end of MEVE, in Figure 16. This stretch of road is a high priority for highway mitigation (i.e., structures to accommodate wildlife passage) for the State of Colorado because of a significant history of wildlife-vehicle collisions (WVC) (Cooley et al. 2020). A recent comprehensive study (Kintsch et al. 2019) identifying wildlife-highway conflict

areas where targeted mitigation could have the greatest impact on reducing WVCs, highlighted this important “hotspot” on Hwy 160 for problematic deer and elk road crossings, particularly during migration season. Kintsch et al. (2019) found that the six miles of Hwy 160 immediately north of the Park ranked in the top 20th percentile for mitigation prioritization for Colorado’s Western Slope roads and highways.

3.3. Rare Plants

MEVE is one of eight hotspots for rare and endemic plants in the state of Colorado (Rondeau et al. 2011). Currently, there are fourteen rare native plants of conservation concern in MEVE. The surveyed distribution of rare species in the Park varies from highly restricted (~1 acre, Giant hellebore [*Epipactis gigantea*]) to fairly widespread (~2000 acres, Chapin Mesa milkvetch [*Astragalus schmollii*]) (Colorado Natural Heritage Program 2023). Of all the rare and important plants in MEVE, the Chapin Mesa milkvetch has been the most heavily monitored. This is because of a prior listing petition for this species, which resulted in a U.S. Fish & Wildlife Service proposal to list Chapin Mesa milkvetch under the Endangered Species Act (ESA) in 2020 (U.S. Fish and Wildlife Service 2022). However, because of the strong conservation measures and implementation plan established in the 2018 Conservation Plan for the Chapin Mesa Milkvetch (Mesa Verde National Park 2018), in 2022 the U.S Fish & Wildlife Service withdrew the proposed rule to list Chapin Mesa milkvetch as a threatened species (U.S. Fish and Wildlife Service 2022).



Mesa Verde stickseed (*Hackelia gracilentia*).

NPS / NIKITA PRASSER

Rare Plants Gap Analysis Summary

The Colorado Natural Heritage Program (CNHP) provided all data utilized in our analyses for rare plants (Colorado Natural Heritage Program 2023). The data consists of polygons representing the surveyed locations of various rare plant species in MEVE (Table 3). CNHP compiled spatial data and metadata from as many reports and journal articles on rare plants within MEVE as possible and created polygons based on that effort. Each polygon represents a distinct survey area for a rare plant species based on its known or suspected occupied habitat within the park, with some survey areas monitored only once and others multiple times. Because the surveys were conducted by different researchers, the survey methods are often inconsistent between polygons. For all rare plants, except the Chapin Mesa milkvetch (CMM) and Cliff Palace milkvetch (CPM) (*Astragalus deterior*), these polygons do not reflect the entire distribution of species within the Park, which are unknown. CMM and CPM are the only species with known full distributions because they are found in smaller distinct locations within MEVE and their habitat is easily accessible for surveyors compared to species such as the Gray's Townsend daisy (*Townsendia glabella*), which is found only on steep slopes (Colorado Rare Plant Guide 2024).

The rare plant surveys span from 1890 to 2021, with the first observation, last observation, and some intermittent observations provided for each polygon. We have divided this dataset into polygons for which the “last observation” was recorded before 2000 (Figure 17a), and polygons for which the “last observation” was recorded after 2000 (Figure 17b). The pre-2000 data is less detailed than the post-2000 data, including less accurate documentation of estimated population sizes and boundaries of surveyed distributions. Therefore, the maps and summaries reported afterwards will only summarize the post-2000 data. The plant polygons provided by CNHP contained attribute data such as approximate density of plants, total acreage represented by polygons, and Element Occurrence “ranks” assigned by CNHP (Colorado Natural Heritage Program 2023) (Figure 17).

Table 3. Summary from the Colorado Natural Heritage Program dataset (Colorado Natural Heritage Program 2023) of rare plants monitored at least once in MEVE after 2000. Any surveys within MEVE not provided to CNHP are not included in this dataset. Approximate plant density within survey polygons values were taken from the latest record assigned to that specific polygon. “Element Occurrence (EO) ranks” were reported within the attribute table of the species’ survey location polygons and are considered to be an expert opinion of “current condition” of each species based on an A–F grade, with a rank of A considered to be healthy and F considered to be extirpated. Because each polygon was reported with a single EO rank, we listed all ranks corresponding with post-2000 polygons for each species. CNHP did not include negative surveys in this dataset.

Scientific name	Common name	Endemic to 4-Corners Region ^A	Global Rank ^B	State Rank ^B	Year of Latest Observation in MEVE	Approximate Plant Density Within Survey Polygons (individuals/acre)	Total Acreage of All Survey Polygons	EO Rank(s)
<i>Acer grandidentatum</i>	Bigtooth maple	–	G4G5T4T5	S1	2020	7	5	B/C
<i>Aletes sessiliflorus</i>	Sessileflower Indian Parsley	X	G3	S2	2018	6	180	A, B, E
<i>Aliciella haydenii</i>	San Juan gilia	–	G3	S2	2020	Unknown	50	A, B
<i>Astragalus deterior</i>	Cliff Palace milkvetch	X	G1	S1	2020	7	280	A, B, C
<i>Astragalus schmolliae</i>	Chapin Mesa milkvetch	X	G1	S1	2021	2	2050	A, B
<i>Collomia grandiflora</i> ^C	Large-flowered collomia	–	G5	S1	2019	615	20	A, E
<i>Epipactis gigantea</i>	Giant hellebore	–	G4	S2	2011	2270	~1	E
<i>Grindelia arizonica</i>	Arizona gumweed	–	G4	S2	2021	65	70	B, E
<i>Hackelia gracilentia</i>	Mesa Verde stickseed	X	G2	S2	2019	40	290	A, B, C, C/D, D
<i>Lepidium crenatum</i>	Alkali pepperweed	–	G3	S3	2018	29	320	A, A/B, E
<i>Townsendia glabella</i>	Gray’s Townsend daisy	–	G2G3	S2S3	2021	3	50	A

^A An "X" indicates presence.

^B G and S ranks sourced from Nature Serve: <https://explorer.natureserve.org/>. This ranking system allows a quick assessment of a species’ rarity, where species are assigned both a global (G) and state (S) rank on a scale of 1 to 5. Both G and S ranks are assigned through a collaborative process involving NatureServe and State Natural Heritage Programs and their scientists.

^C MEVE’s population of *Collomia grandiflora* is disjunct from the rest of the range of the species, which is common outside Colorado. Because of this, additional genetics research may be needed to ascertain whether the MEVE population might be distinguished as a new species.

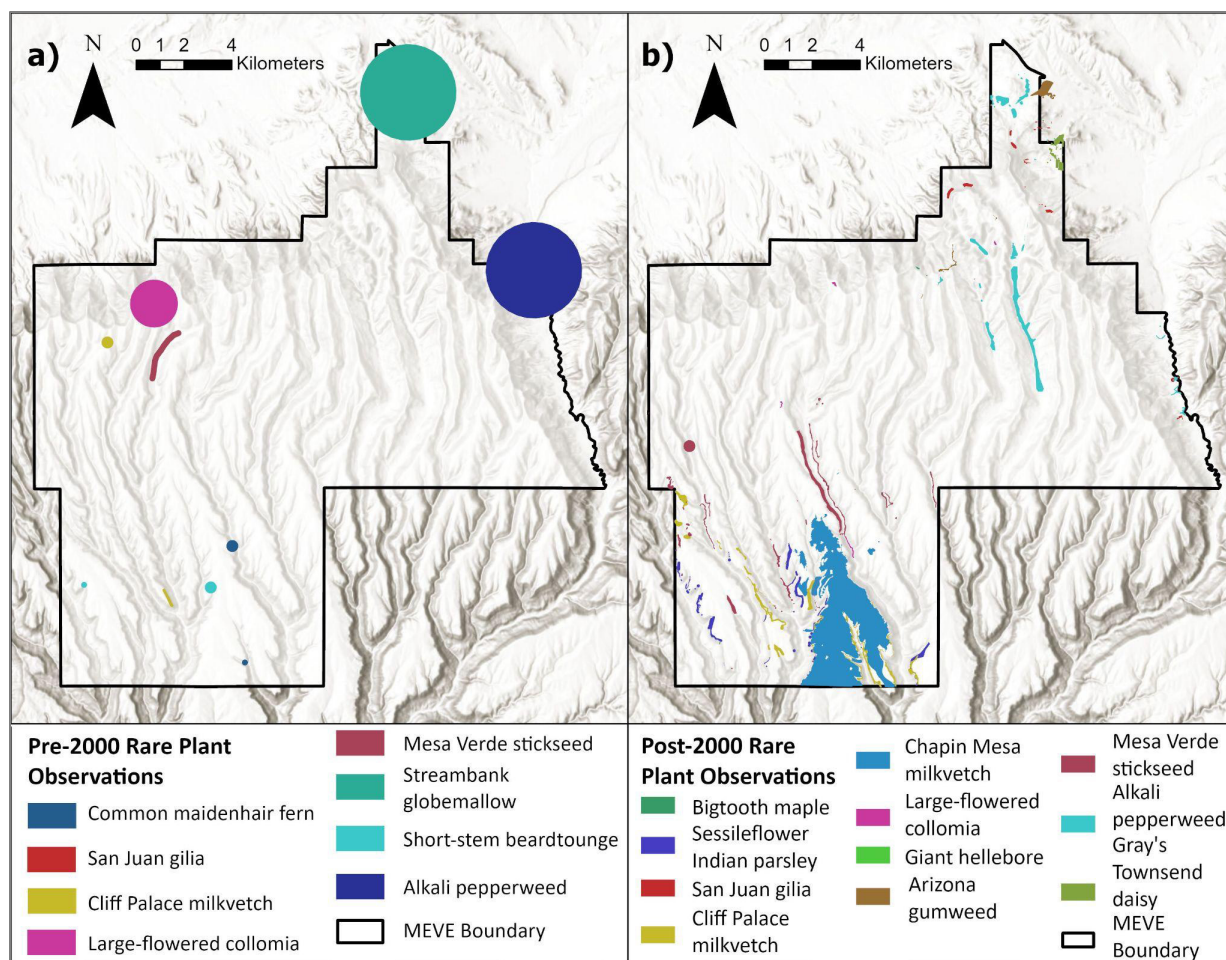


Figure 17. Surveyed locations of 14 species of rare plants within MEVE. a) Documented rare plant locations last surveyed before 2000. b) Documented rare plant locations last surveyed after 2000. Created using data from the Colorado Natural Heritage Program (2023).

NPS / CONSERVATION SCIENCE PARTNERS

We overlaid historical fire perimeters from 1970 and onward onto the rare plant polygons to understand the degree of fire disturbance within the surveyed rare plant locations in MEVE (Figure 18). Rare plant species that overlapped with post-1970 fires within MEVE (larger than 10 acres) included bigtooth maple, Sessileflower Indian parsley, San Juan gilia, Cliff Palace milkvetch, Chapin Mesa milkvetch, large-flowered collomia, Arizona gumweed, Mesa Verde stickseed, and alkali pepperweed. Approximately 17% (570 out of 3,300 acres) of these rare plants' survey locations overlapped with post-1970 fire. This is less overlap than expected, but as the complete survey distributions are unknown for all rare species, other than CMM and CPM, we cannot determine if fire has had little effect on these plants without more information on their full whereabouts.

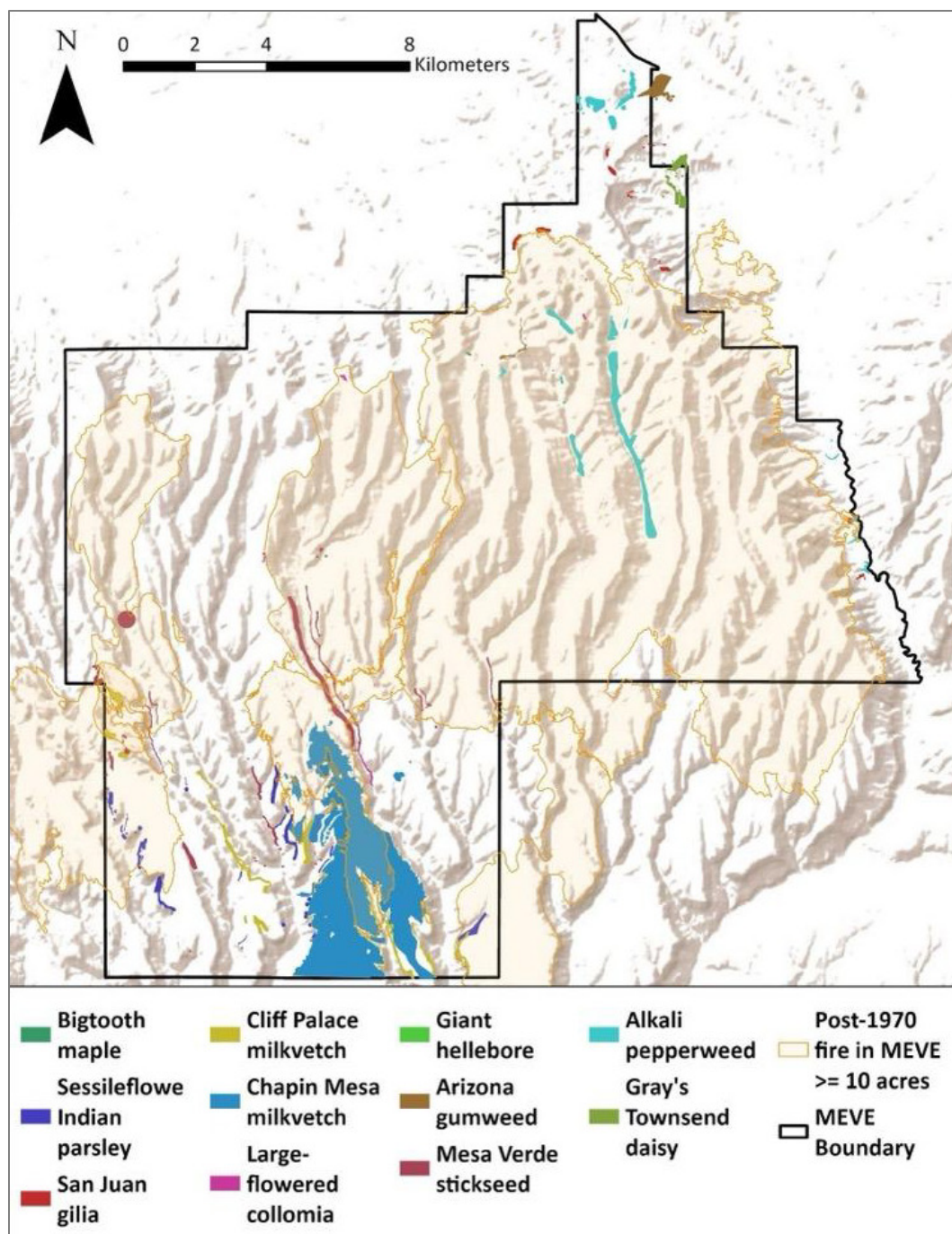


Figure 18. Survey locations of 11 species of rare plants within MEVE overlaid with historical fires ≥ 10 acres from 1970 and onward. Created using rare plant data from the Colorado Natural Heritage Program (2023) and fire data from the National Interagency Fire Center (2022).

NPS / CONSERVATION SCIENCE PARTNERS

We analyzed the percent cover of annual grasses and forbs in the Park to determine the extent of non-native species within rare plant surveyed distributions (Figure 19). We used a dataset from the RAP from 2023 that recorded the percent cover of annual grasses and forbs across the contiguous United

States (30 m resolution), with each pixel ranging in value from 0 to 100% (Allred et al. 2021). We found that Colorado had a range of 0–100% annual grass and forb coverage per pixel, with an average of $5.62\% \pm 8.79\%$, while MEVE had a range of 0–46%, with an average of $3.73\% \pm 4.08\%$. We then examined the percent cover of annual grasses and forbs within each rare plant species survey polygon to determine which rare plants were potentially most impacted by non-native species (Table 4). The survey polygons of Arizona gumweed, Chapin Mesa milkvetch, and Alkali pepperweed had the greatest percent cover of annual forbs and grasses (8.67%, 4.81%, and 4.07%, respectively), while large-flowered collomia, giant hellebore, and Gray’s Townsend daisy had the lowest (1.47%, 1.50%, and 1.76%, respectively). Not all rare plant species are negatively impacted by non-native annual forbs and grasses. When making management decisions regarding a particular species, we recommend readers conduct a literature review to understand the needs of the species and if the presence of annual grasses and forbs is an issue for the plant.

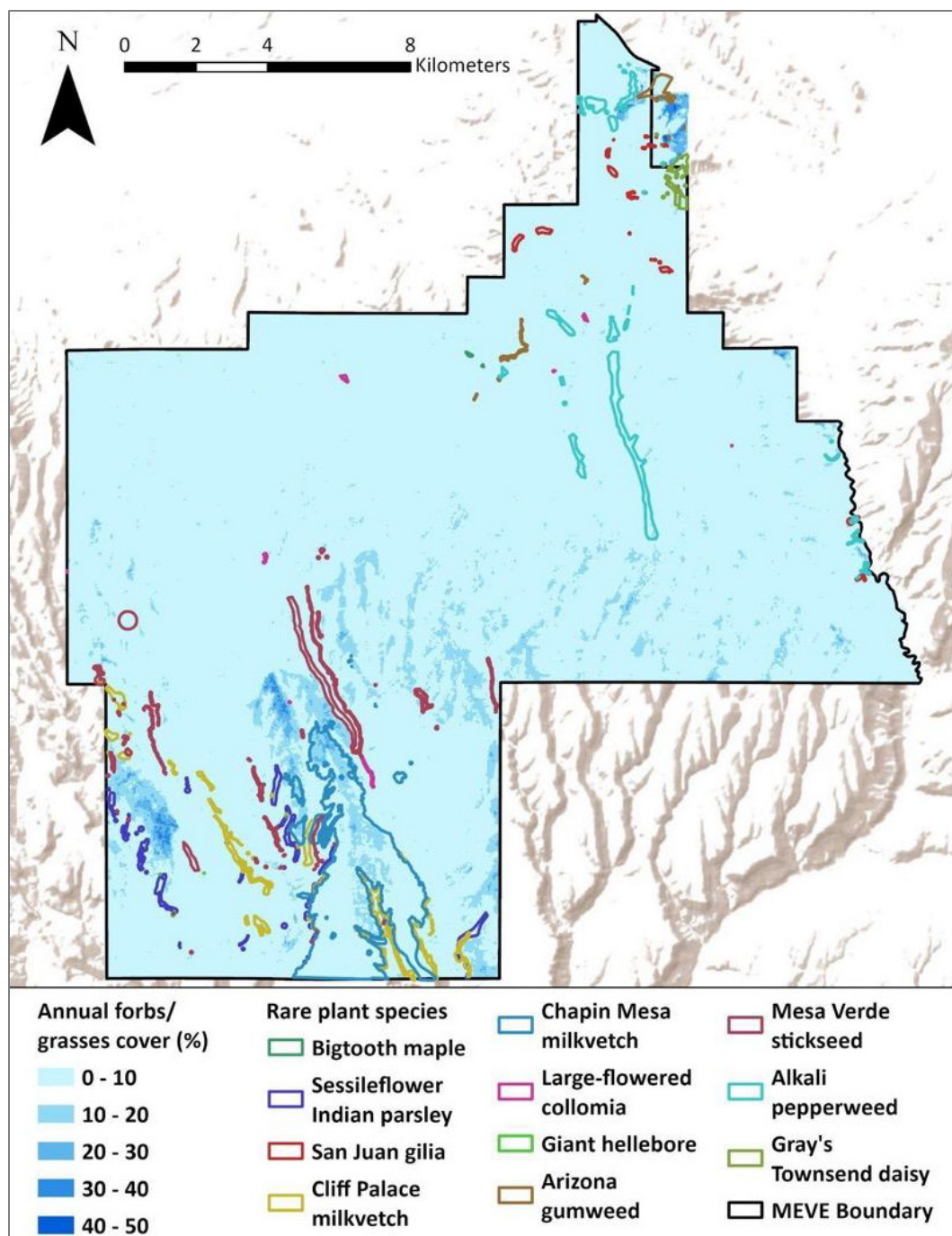


Figure 19. Survey locations of 11 species of rare plants within MEVE overlaid with the percent cover of annual grasses and forbs in 2023. Created using rare plant data from the Colorado Natural Heritage Program (2023) and annual forbs and grass data from the Rangeland Analysis Platform (Allred et al. 2021).

NPS / CONSERVATION SCIENCE PARTNERS

Table 4. The percent cover of annual grasses and forbs (Allred et al. 2021) from RAP 2023 within the surveyed distribution of 11 species of rare plants in MEVE. Min and max refer to the lowest and highest pixel values within each species' survey polygons. The mean is the average pixel value across all polygons of each species.

Species name	Common name	Min	Max	Mean	Standard Deviation
<i>Acer grandidentatum</i>	Bigtooth maple	0.00	3.00	2.00	0.95
<i>Aletes sessiliflorus</i>	Sessileflower Indian parsley	0.00	18.00	2.59	2.79
<i>Aliciella haydenii</i>	San Juan gilia	0.00	13.00	2.47	1.96
<i>Astragalus deterior</i>	Cliff Palace milkvetch	0.00	21.00	3.92	2.88
<i>Astragalus schmolliae</i>	Chapin Mesa milkvetch	0.00	33.00	4.81	5.90
<i>Collomia grandiflora</i>	Large-flowered collomia	0.00	9.00	1.47	1.52
<i>Epipactis gigantea</i>	Giant hellebore	1.00	2.00	1.50	0.50
<i>Grindelia arizonica</i>	Arizona gumweed	0.00	41.00	8.67	7.21
<i>Hackelia gracilentia</i>	Mesa Verde stickseed	0.00	24.00	4.06	3.26
<i>Lepidium crenatum</i>	Alkali pepperweed	0.00	39.00	4.07	3.82
<i>Townsendia glabella</i>	Gray's Townsend daisy	0.00	20.00	1.76	2.27

Cliff Palace Milkvetch Gap Analysis Summary

Cliff Palace milkvetch (*Astragalus deterior* [Barneby]) (CPM) has been surveyed and monitored more often than the other rare plants tracked by MEVE along with CMM. CPM is a short-lived perennial forb endemic to the Mesa Verde cuesta (Figure 20). It is found in shallow sandy soils along the rocky rims of mesas and slopes with deposits from the Upper Cliff House Formation.

Documented threats to CPM include trampling by visitors on trails, impacts by park infrastructure (roads, waterlines), erosion, non-native species and climate change (Moore 2008).

CPM's population was partially burnt in the Pony Fire of 2000 and Long Mesa Fire of 2002 (Figure 20). In 2003, Moore (2008) established 11 permanent transects for MEVE's CPM population, representing both recently burned and unburned areas. CPM was then repeatedly monitored in 2004, 2005 and 2007. In the summer of 2021, the CPM plots were found and revisited (Stowell 2022a). The protocols used in 2021 attempted to replicate those used by Moore (2008). In 2022, new larger belt transects were established in the same areas where the 2003 transects were located (Stowell 2022a), while retaining similar protocols to those used by Moore (2008).

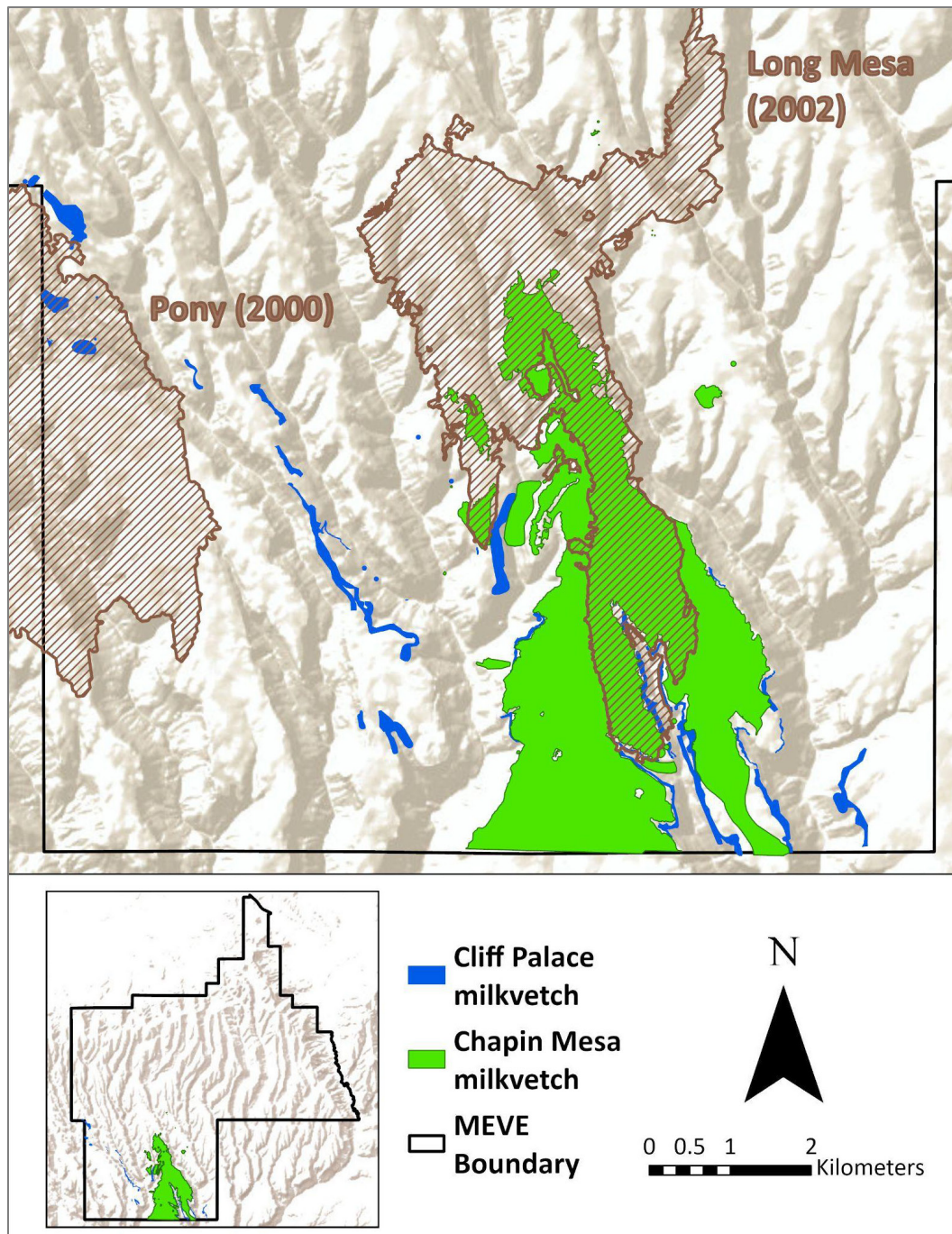


Figure 20. Known range of Cliff Palace milkvetch (CPM) and Chapin Mesa milkvetch (CMM) within MEVE. The footprints of the Pony Fire (2000) and Long Mesa Fire (2002) are also displayed. Created using rare plant data from the Colorado Natural Heritage Program (2023) and fire data from the National Interagency Fire Center (2022).

NPS / CONSERVATION SCIENCE PARTNERS

We compared the density of CPM in the earlier 2000s to that of the 2020s in burned and unburned transects (Figure 21a). The average density of CPM in burned transects was 0.82 individuals/m² in

the 2000s and 0.05 individuals/m² in 2021 and 2022, representing about a 95% decline. The average density of CPM in unburned transects was 0.84 individuals/m² in the 2000s and 0.20 individuals/m² in 2021 and 2022, representing about an 80% decline. Although CPM density in the unburned transects was higher than that of the burned transects in 2021–2022, the major decline in abundance across both transects from the 2003–2007 estimates is cause for concern. This suggests that there may be another threat influencing population dynamics and driving decline, such as warmer temperatures and drought due to climate change. Five of the previous 10 years prior to monitoring in 2021 and 2022 displayed below average precipitation (Figure 21b). Below average precipitation for multiple years may have led to a lack of CPM seedlings recruiting to adults and some CPM adults prematurely dying before reproducing (personal communication, Tova Spector, US Forest Service, October 2024).

We compared seedling counts between the early 2000s and early 2020s. During the early 2000s, 414 seedlings were counted in burned and unburned transects combined (Moore 2008). When averaged across those four years, the percentage of all individuals counted that were seedlings at burned transects was 24%, and the percentage of seedlings (of all CPM individuals counted) at unburned transects was 36%. For the early 2020s, only seedling counts for 2022 were available for comparison to those from the early 2000s. Out of the 10 transects from the early 2000s that were resurveyed in 2022, nine of those transects recorded 0 seedlings (Stowell 2022a), again illustrating a dramatic decline in abundance of seedlings from 2007 to 2022 regardless of whether the plot was in the burned or unburned area.

In summary, CPM depends on seed production and banking to maintain a stable and viable population. Because CPM has a low density in both burned and unburned transects as of recent years, seed production is likely limited thus contributing to continued decline in both (personal communication, Tova Spector, US Forest Service, August 2024). Moreover, a genetic analysis conducted by McCauley and Cortes-Palomec (2023) found that the small population size and habitats of CPM is leading to inbreeding. Notably, CPM appears to be dramatically declining across both burned and unburned habitat. This could be due to warming temperatures and drought caused by climate change, and Park activities such as trampling and other disturbance could be further exacerbating this stressor (personal communication, Tova Spector, US Forest Service, September 2024). Monitoring and protective measures should continue and be expanded for this species, as the apparent decline of this population warrants immediate Park management action to ensure this species does not become extirpated from MEVE.

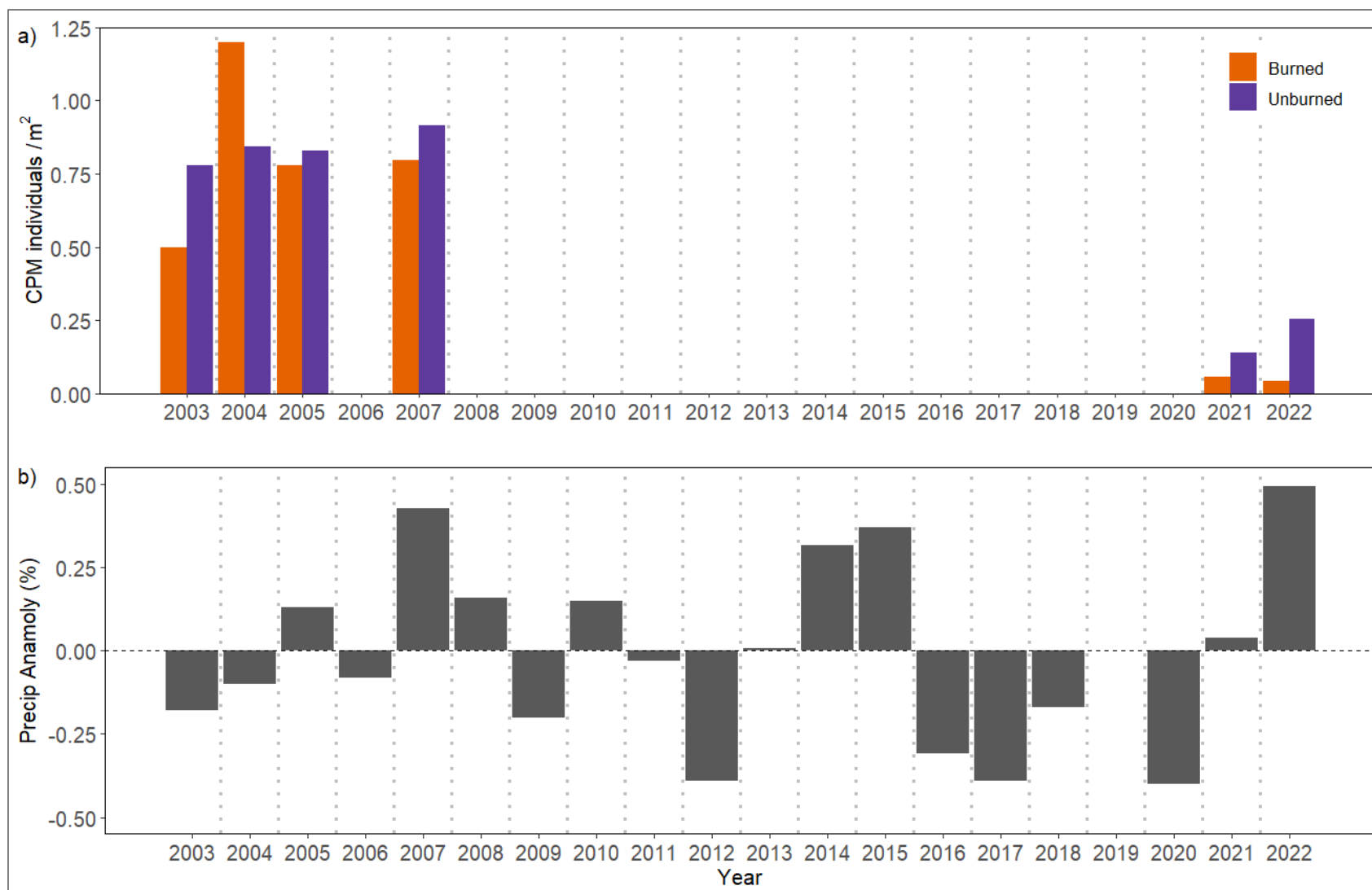


Figure 21. a) Density of CPM (individuals / m²) in 2003, 2004, 2005, 2007, 2021 and 2022 in burned and unburned transects. b) MEVE precipitation deviation from 2003 to 2022. Created using rare plant data from MOORE (2008) and STOWELL (2022A) and precipitation data from the National Weather Service (N.D.).

NPS / CONSERVATION SCIENCE PARTNERS

Chapin Mesa Milkvetch Condition Assessment Summary

The Chapin Mesa milkvetch (*Astragalus schmolliae*) (CMM) is by far the most studied plant in MEVE. CMM is endemic to the Chapin Mesa of MEVE, with some small outlying populations on Park Mesa (see Figure 20). The species is found within intact PJ woodlands, with old growth PJ the preferred habitat, (Rondeau et al. 2022), and is dormant during winter, not sprouting any above-ground stems during that season. Dormancy outside of winter months occurs when there is a lack of deep soil moisture (35–45 cm soil depth). After severe winter droughts, CMM remains dormant during the growing season (spring-fall), as winter precipitation is necessary to replenish deep soil moisture (Rondeau 2019; Rondeau et al. 2022).

Friedlander (1980) was the first to provide population estimates of CMM within MEVE. Further monitoring of CMM occurred in 2001, 2002, 2003, and 2011–2023 by MEVE Park staff and other researchers. In 2002, the Long Mesa Fire burned 38% of the area known to be occupied by CMM (Anderson 2004) (Figure 20). The CMM within the burned areas is now considered a separate subpopulation from that within the unburned areas (personal communication, Renee Rondeau, CO Natural Heritage Program, December 2023).

After the 2002 fire, Chapin Mesa burned areas were seeded with native bunch grasses including western wheatgrass (*Pascopyrum smithii*). These grasses were seeded specifically to prevent the intrusion of non-native cheatgrass and other non-native forbs, such as musk thistle. From 2015–2022, cheatgrass (*Bromus tectorum*) decreased by ~35% and western wheatgrass increased by ~25% within the burned plots (Rondeau 2023). Post-seeded western wheatgrass increased more in the burned areas than it naturally would have and competed with all age classes of CMM, further threatening CMM population viability in burned areas.

Renee Rondeau of the Colorado Natural Heritage Program (CNHP) provided all data utilized in our analyses for CMM. These data include those collected from the most recent survey period (2011–2023) using established, permanent transects that cover the majority of Chapin Mesa within the Park. Following Rondeau et al. (2023), we considered dormant CMM when analyzing adults or immatures. By factoring in dormancy, researchers avoid assuming that if a CMM individual is missing a certain year, it must be dead, which is important given that dormancy is a major life history trait of CMM. Researchers began calculating CMM dormancy beginning in 2015; therefore, we were unable to utilize data from 2011 to 2014 for CMM population trends. When reporting dormancy, we did not utilize 2022–2023 data because of the likelihood of a two-year dormancy for some individuals (dormant plants would not be “captured” in the count without a two-year buffer). Dormancy does not apply to seedlings or yearlings.

Population trends. Before the 2002 Long Mesa Fire, CMM density of emerged individuals was similar in to-be-burned (0.36 individuals/m²) and unburned (0.40 individuals/m²) transects on Chapin Mesa (Rondeau 2019) (Figure 22). For several years after the Long Mesa Fire, the relative density of emerged individuals in burned transects was greater than unburned transects (Figure 22). Anderson (2004) attributed these higher CMM densities in burned transects to the lack of heterospecific competition for resources such as water, sunlight, nutrients and pollinators.

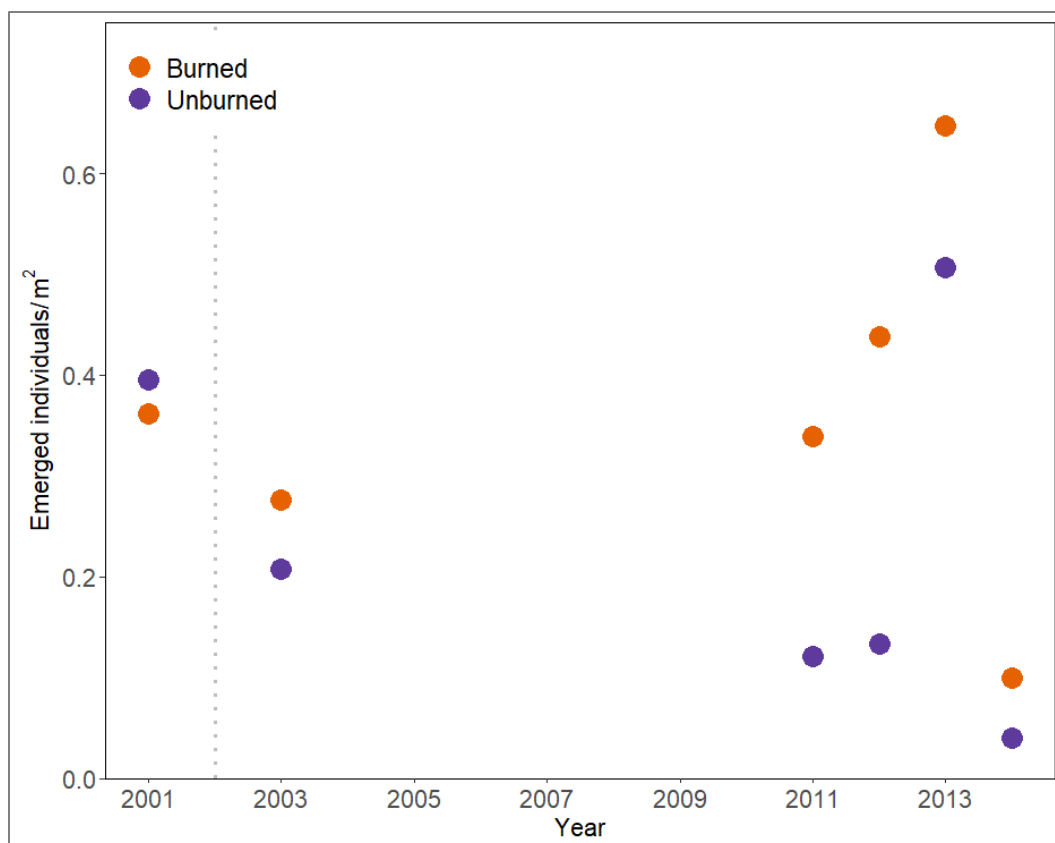


Figure 22. Density (individuals / m²) of emerged CMM individuals of all age classes in 2001, 2003, and 2011 to 2014 in burned and unburned transects. The dotted gray line represents the year of the Long Mesa Fire (2002). Created using rare plant data from Renee Rondeau of the Colorado Natural Heritage Program.

NPS / CONSERVATION SCIENCE PARTNERS

To analyze trends of emerged and dormant CMM adults and immatures of both subpopulations from 2015 to 2021, we utilized a Generalized Least Squares (GLS) regression model as it accounts for non-independence between observations, as found in this dataset. There was a significant effect of year on the density of emerged and dormant CMM adults and immatures in the burned population ($\beta_1 = -0.0146$, p-value = 0.0410), indicating a decreasing trend in density of these age classes over time in this population (Figure 23). For the unburned population, there was not a significant effect of year on density ($\beta_0 = -18.1191$, p-value = 0.2713), suggesting the density of emerged and dormant adults and immatures in this subpopulation were stable (Figure 23). Assumptions were tested and residuals were normally distributed via a Shapiro test (p-value > 0.05) and no significant autocorrelation was detected (Ljung-Box test p-value > 0.05). Further details on the methodology for fitting GLS models can be found in Appendix B.2.

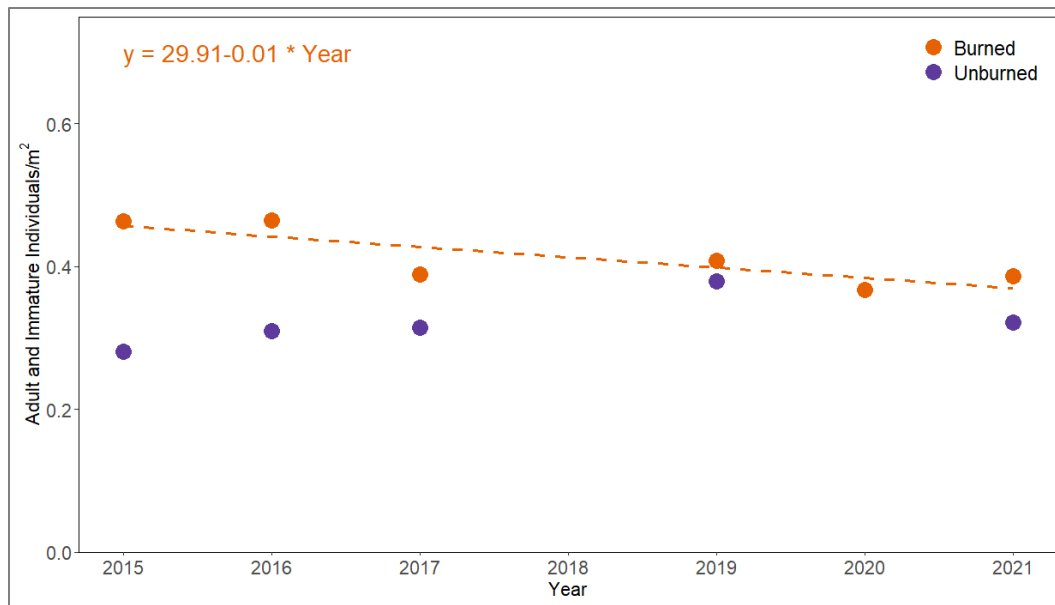


Figure 23. Density (individuals / m²) of adult and immature (emerged and dormant) CMM from 2015 to 2021 in burned and unburned transects. The GLS model's fitted values from the burned population are represented as a trendline. The associated equation of the trendline is also reported. The unburned population's GLS model was not reported. Created using rare plant data from Renee Rondeau of the Colorado Natural Heritage Program.

NPS / CONSERVATION SCIENCE PARTNERS

Rondeau et al. (2022) suggested the presence of PJ woodlands in the unburned landscape may be beneficial to that CMM subpopulation as it causes low heterospecific competition for soil moisture, adequate pollinators, and ample space for CMM seed germination. This would account for the stability in older age classes of this subpopulation from 2015 to 2021. However, for the burned CMM subpopulation, a landscape that once lacked competitors 20 years ago is now inundated with non-native grasses. Rondeau et al. (2022) proposed that a downward trend of CMM density in the burned subpopulation was due to increased competition from other herbaceous species, as well as increased temperatures, increased sunlight, and reduced pollinators. However, even with a downward trend, the burned population's density of adults and immatures was consistently higher than the unburned population during 2015–2021 (Figure 23).

CMM Demography

In favorable years, CMM fruits are present in late May and expel their seeds in late June (Rondeau et al. 2022). CMM seeds are long-lived and can withstand fires. Because seeds are not counted during monitoring, seed pods, i.e., fruit, are counted instead. There was little difference between the two populations regarding this metric, as fruit density in unburned transects was higher than in burned transects only 5 out of 9 years (Figure 24a). From 2015 through 2022, fruit density remained below 5 fruits/m². However, in 2023, there was a spike in fruits recorded in both unburned and burned transects (~17 and 13 fruits/m², respectively).

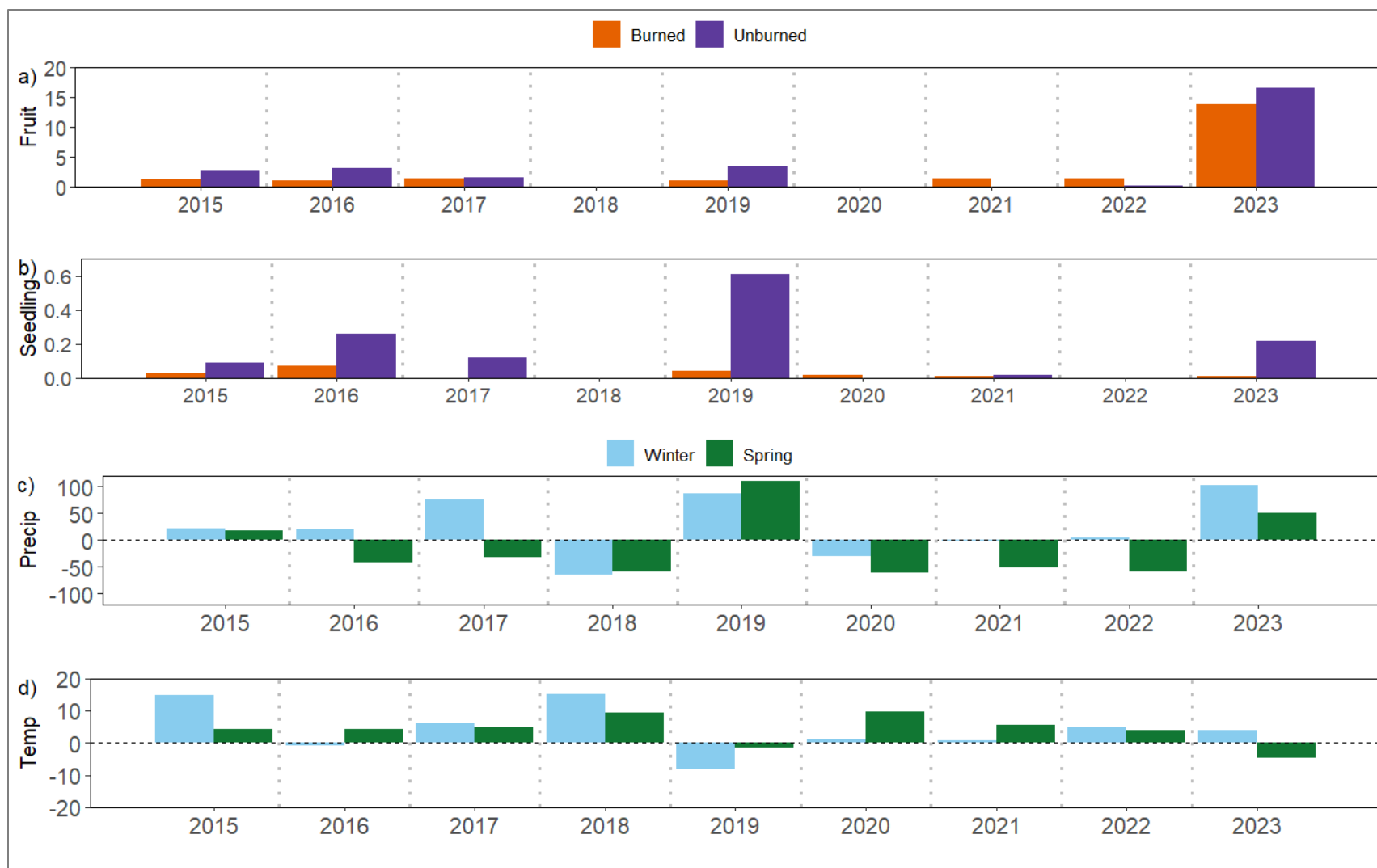


Figure 24. Plots of a) CMM fruit density (fruit/m²), b) CMM seedling density (seedlings/m²), c) MEVE precipitation anomalies (%) (winter and spring), and d) MEVE temperature anomalies (%) (winter and spring) from 2015 to 2023. Created using fruit and seedling data from Renee Rondeau of the Colorado Natural Heritage Program and precipitation and temperature data from the National Weather Service (N.D.).

NPS / CONSERVATION SCIENCE PARTNERS

Fruit density was lower than expected in 2019 despite higher than average precipitation (Figure 24a and c). To better understand the impacts of precipitation and temperature on fruit density and resulting reproductive output, we ran multiple Generalized Least Squares (GLS) regression models to predict fruit density from temperature and precipitation variation (winter and spring) from 2015 to 2023, considering multiple combinations of the temperature and precipitation variables. All models included a correlation structure accounting for autocorrelation between yearly observations. Precipitation deviations were calculated from 1971 to 2000 winter and spring averages, and temperature deviations were calculated from 1970 to 2000 winter and spring averages (National Weather Service, n.d.). We combined fruit density data from burned and unburned transects because there was little difference in interannual variability between the two populations, so both were likely similarly impacted by environmental variation. This combined fruit density was then log-transformed. The model with spring temperature deviation alone was the best supported candidate model based on AIC and indicated a negative relationship between temperature deviation and fruit density ($\beta_1 = -0.1648$, $p\text{-value} = 1e-04$). Assumptions were tested and residuals were normally distributed via a Shapiro test ($p\text{-value} > 0.05$) and no significant autocorrelation was detected (Ljung-Box test $p\text{-value} > 0.05$). Further details on the methodology can be found in Appendix B.3.

CMM is entomophilic, which means they rely on insect pollinators to produce fruit and ultimately, seeds. Therefore, our findings that high fruit density correlated with lower spring temperatures would suggest that CMM pollinators, such as bees, fare better in these spring conditions, assuming spring temperatures are not so low that flowers freeze. With such high fruit density in 2023, the current conditions for this metric are hopeful in both burned and unburned transects. However, things may quickly change if MEVE experiences drier and hotter spring conditions as predicted by National Park Service (2023) climate scenarios. Hotter spring conditions may not directly negatively affect CMM because the species has deep roots that reach soils replenished by winter moisture (personal communication, Renee Rondeau, CO Natural Heritage Program, August 2024). However, these spring conditions would negatively affect pollinators who are important to CMM fruiting and subsequent seeding.

Seedling density varied across time due to environmental impacts, with densities as low as 0 individuals/m² during a drought (2018) and as high as 0.6 individuals/m² during wet spring conditions in the unburned population (2019) (Figure 24b and c). Every year when seedlings were observed, unburned transects had a higher density than burned transects. Because fruit production was similar between burned and unburned transects (Figure 24a), the lack of seedlings within the burned areas was likely due to competition from grasses, especially western wheatgrass (Rondeau 2023).

To better understand the impacts of precipitation and temperature on seedling density, we ran multiple Generalized Least Squares (GLS) regression models to predict seedling density from temperature and precipitation variation (winter and spring) from 2015 to 2023, considering multiple combinations of the temperature and precipitation variables. Precipitation deviations were calculated from 1971 to 2000 winter and spring averages, and temperature deviations were calculated from 1970 to 2000 winter and spring averages (National Weather Service n.d.). We utilized only unburned

seedling density because of the lack of seedlings within burned transects. The model with spring precipitation deviation alone was the best supported candidate model based on AIC and indicated a negative relationship between precipitation deviation and seedling density ($\beta_1 = 0.0028$, $p\text{-value} = 0.0083$). Assumptions were tested and residuals were normally distributed via a Shapiro test ($p\text{-value} > 0.05$) and no significant autocorrelation was detected (Ljung-Box test $p\text{-value} > 0.05$). Further details on the methodology can be found in Appendix B.3.

The correlation between spring precipitation and seedling density supports the observation that seedling emergence requires a wet spring (Rondeau 2019). Seedling density was noticeably lower in 2020–2022 than in previous years, especially for unburned CMM, due to low precipitation (Figure 24b and c). In 2023, seedling density rebounded in the unburned transects due to higher-than-average precipitation (Figure 24b and c) but remained poor in the burned transects. For seedlings to survive into subsequent age classes (yearlings and immatures), they need two consecutive winters with near-average or above-average moisture levels (Rondeau 2019). From 2015–2021, seedling survival was ~8% in the burned areas and ~7% in the unburned areas, with a 6% threshold considered as “good” (personal communication, Renee Rondeau, CO Natural Heritage Program, April 2024). A future climate of drier spring conditions as predicted by the National Park Service (2023) climate scenarios may lower CMM seedling emergence and survival. CMM is a long-lived perennial (at least 50 years), and episodic seedling years are normal and not of concern (personal communication, Renee Rondeau, CO Natural Heritage Program, August 2024). However, if the climate shifts so that all springs are dry, CMM may be unable to maintain a viable population.

To ensure long term longevity of any population, it is important that age class distributions are not disproportionately skewed to any one class (i.e., a population where almost all individuals are mature, but with hardly any seedlings or yearlings). We assessed if there were significant differences between four CMM age class densities (individuals/m²) (adults [emerged and dormant], immatures [emerged and dormant], yearlings [one-year old plant], and seedlings) in burned and unburned transects pooled across 2015–2021 (Figure 25) using a Kruskal-Wallis test. Overall density was typically higher within unburned transects compared to the burned transects (Figure 25). Within the burned transects, the density of adults significantly differed from immatures, seedlings, and yearlings ($p\text{-values} = 0.042, 0.013, \text{ and } 0.011$, respectively). There was also a significant difference in immatures compared to yearlings ($p\text{-value} = 0.040$) but not seedlings ($p\text{-value} = 0.090$). There was also no significant difference in seedlings and yearlings ($p\text{-value} = 1$). Unburned transects had zero significant differences between age classes ($p\text{-values} > 0.05$). Further details on the methodology can be found in Appendix B.3.

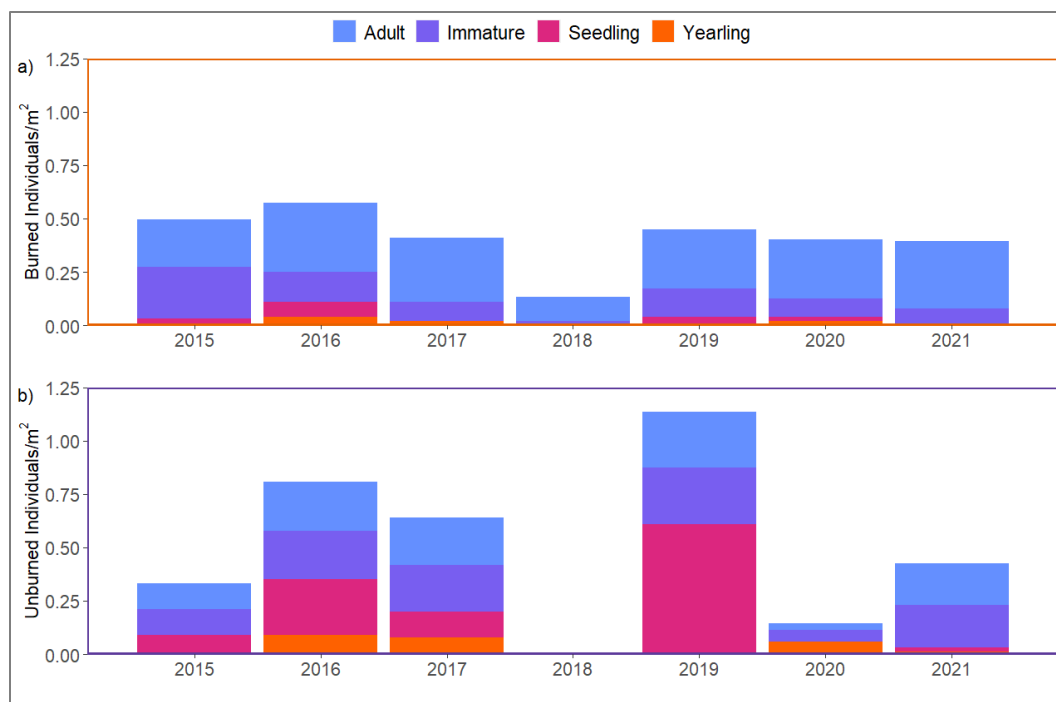


Figure 25. Distribution of CMM age class densities from 2015 to 2021 in a) burned and b) unburned transects. Age classes consist of adults (emerged and dormant), immatures (emerged and dormant), seedlings (emerged), and yearlings (emerged). Unburned transects in 2018 were surveyed; however, no plants emerged, due to the severe winter drought. Created using rare plant data from Renee Rondeau of the Colorado Natural Heritage Program.

NPS / CONSERVATION SCIENCE PARTNERS

The unburned CMM population had a more balanced distribution of age classes compared to the burned CMM population. With significantly higher densities of younger age classes as well, we inferred that the unburned CMM population experienced greater recruitment success than the burned population. This is important because, for the long-term persistence of the Park’s CMM population, mortality must be offset by recruitment. This is of great concern for the burned population, as the low recruitment in burned areas due to the lack of seedlings and the survival of the few emerged seedlings (Figure 24b) has created an evident shift towards older age classes (Figure 25a), which were also significantly trending downwards (Figure 23). This supports the theory of Rondeau (2023) that CMM in the burned areas will essentially disappear by 2040, other than in germination sites sheltered under shrub and grass cover where CMM recruitment is higher.

To complete the condition assessment for CMM, the current condition status of indicators of population trend, demography, recruitment and seedling survival for both burned and unburned populations were assessed using the condition rating thresholds developed in Appendix A.2. Current density of adults and immatures was not selected as one of the indicators of current condition for CMM. Figure 23 indicates that the densities of burned and unburned populations are currently similar- thus perhaps implying “good” condition. However, with a very long-lived species, such as CMM, current density is not as important as potential future density, which relies on successful

fruiting, seedling establishment, and recruitment over the coming decades. Table 5 below illustrates the outcome of this exercise, based on the data and analyses for CMM presented above.

Table 5. Chapin Mesa milkvetch (CMM) condition ratings summary. Condition ratings below are based on the Condition Rating Table in Appendix A.

Indicator	Condition	Trend	Condition Evaluation
Population trend (unburned)	Good	Stable	This condition is good because the unburned CMM population of adults and immatures was stable (as determined with a GLS regression model), and no mortality was present.
Population trend (burned)	Poor	Downward	This condition is poor because the trend of burned CMM population of adults and immatures was significantly negative (as determined with a GLS regression model).
Demography (unburned)	Good	N/A	This condition is good because there were no significant differences between age class distributions within the unburned population, as determined with a Kruskal-Wallis test.
Demography (burned)	Fair/Poor	N/A	This condition is fair/poor because there were four incidences of significant differences between age class densities. In particular, there were higher densities of older age classes compared to younger. This was determined via a Kruskal-Wallis test.
Fruit production (unburned)	Good	N/A	This condition is good because there was a high density of fruits ($>10/\text{m}^2$) as of the most recent 2023 data collection.
Fruit production (burned)	Good	N/A	This condition is good because there was a high density of fruits ($>10/\text{m}^2$) as of the most recent 2023 data collection.
Seedling density (unburned)	Good	N/A	This condition is good because there was at least 1 year of good seedling density (>0.45 seedlings/ m^2) within the past 10 years of data collection.
Seedling density (burned)	Poor	N/A	This condition is poor because there were no years of good seedling density (>0.45 seedlings/ m^2) within the past 10 years of data collection.
Seedling survival (unburned)	Good	N/A	This condition is good because the unburned seedling survival (7%) was above the threshold for a stable population (6%).
Seedling survival (burned)	Good	N/A	This condition is good because the unburned seedling survival (8%) was above the threshold for a stable population (6%).

3.4. Mancos River (Water Quality, Macroinvertebrates, Fish, Riparian Corridor)

The Mancos River is a second order river and tributary to the San Juan River. It flows south through approximately 8 km (~5 miles) of eastern MEVE on its way to the Ute Mountain Ute Reservation. The river and its riparian forest corridor provide unique habitat for many species in the Park, and also offer important ecological functions. These include but are not limited to contributing to floodplain complexity, channel stability, and providing shade via overhanging vegetation, thus reducing stream

temperature and providing cover for fish. Riparian plant communities range from wet meadows to deciduous forests containing stream-side facultative species such as willows (*Salix* spp.). Because the reach of the river within MEVE is only a small portion of the Mancos River, there are negative influences from upstream land use activities, including agriculture, livestock grazing, dewatering, and potential impacts of historic mining in the watershed (Stacey 2007).



Fall colors along the Mancos River.

NPS / SCPN

This section of the Mancos River is designated by the State of Colorado as a Class 1–Warm Water Biota stream (Colorado Parks and Wildlife 2022). Based on regular surveys of the Mancos River fish fauna spanning at least the last 50 years, 21 species are present or occurred historically in the river (National Park Service 2024). Of these, seven are native species. The bluehead sucker (*Catostomus discobolus*), flannelmouth sucker (*Catostomus latipinnis*), and roundtail chub (*Gila robusta*)¹ are native species of special conservation status (Utah Department of Natural Resources 2006) and are also listed as Species of Greatest Conservation Need (Tier I) in CPW’s State Wildlife Action Plan (Colorado Parks and Wildlife 2015). Because of these three special status native fishes, CPW manages this stretch of the Mancos as a “category 204 or Native Fish Conservation Stream”, a designation which directs CPW to manage for conservation and recovery of native fishes, including management activities like stocking native species and removing non-native predator or competitor species (personal communication, Jim White, Colorado Parks and Wildlife, June 2024).

¹ There still remains today a chance that roundtail chub in the Mancos river retain genetic contamination from previous (2018) accidental stocking of roundtail-Rio Grande chub hybrids into the river.

Condition Assessment Summary for Water Quality

The Southern Colorado Plateau Network (SCPN) provided us with data on the water quality of the Mancos River from a site on the east side of MEVE downstream of where the river enters the park (Southern Colorado Plateau Network 2024). We calculated the average annual values of all water chemistry and core parameters (temperature, pH, and dissolved oxygen) from 2009 to 2021 (Figures 26 and 27). We then compared all averaged annual values to the state standards for Colorado set by the Water Quality Control Commission (Colorado Department of Public Health and Environment, Water Quality Control Commission n.d.). Further details on the methodology can be found in Appendix B.4. Based on available data, all inorganic/non-mineral water quality indicators (pH, dissolved oxygen and non-winter temperatures) were within Colorado's water quality standards across all 12 years of collection (Figure 26). All nutrient indicators (i.e., Nitrite, Ammonia) were within state standards per the most recent data collection (2021) and across all 12 years of collection (Figure 27a, b, and c). All trace metals and mineral indicators (Iron, Aluminum, Cadmium, Lead, Manganese, Molybdenum, Uranium, Boron, Arsenic, Copper, Nickel, Selenium, Zinc, Mercury and Chromium) were within state standards per the most recent data collection (2021) (Figure 27d, e, f, g, h, i, j, k, l, m, n, o, p, q, and r). However, certain trace metals and minerals regularly or occasionally exceeded state standards across our monitoring window. For example, copper exceeded state standards for nine years out of 12 years of data collection (Figure 27m), selenium for one year (Figure 27o), iron for one year (Figure 27d), and aluminum for one year (Figure 27e).

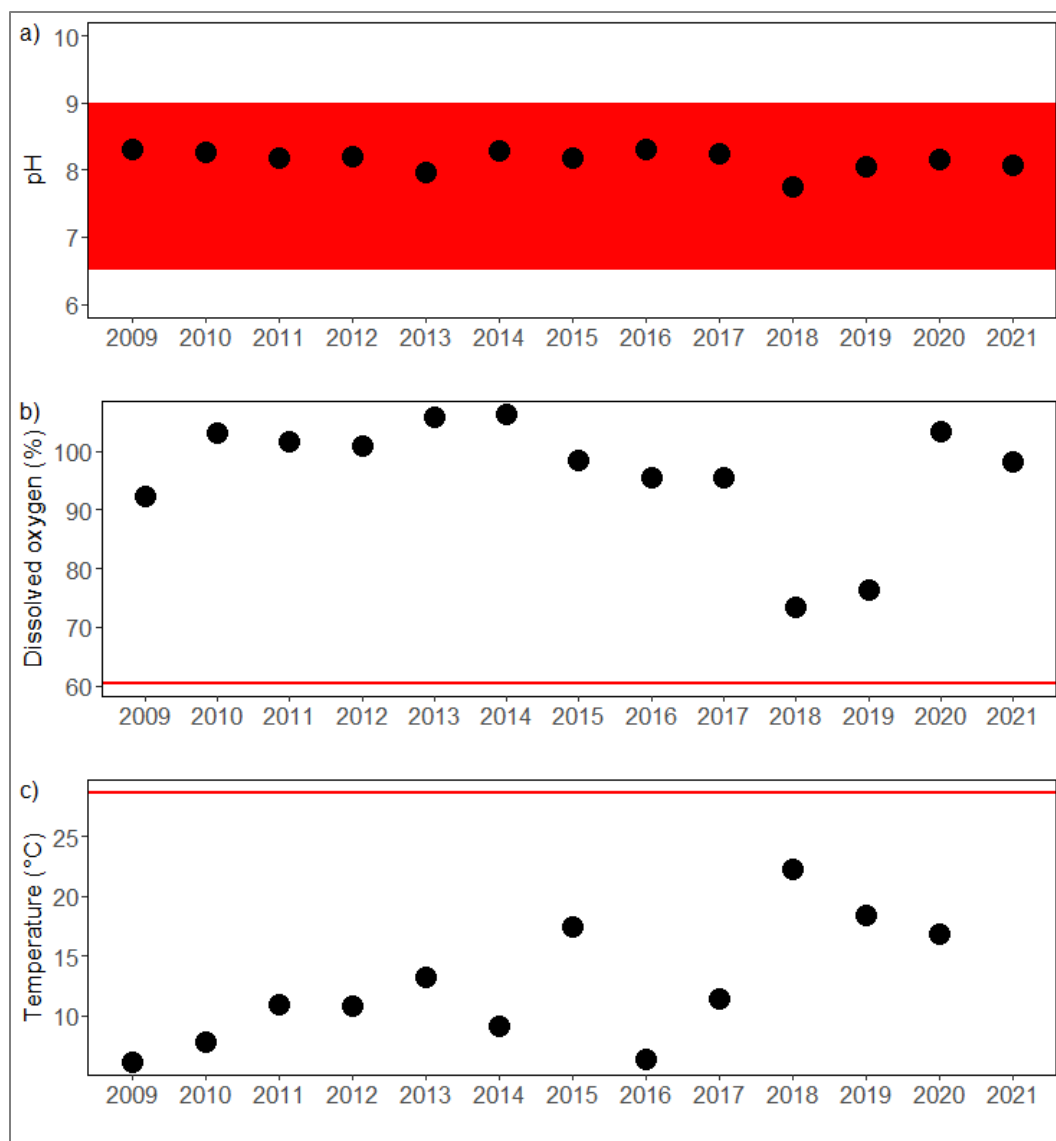


Figure 26. Average annual values of a) pH, b) dissolved oxygen (%) and c) non-winter temperatures (°C) (March–November) from 2009 to 2021. Red polygon in (a) reflects the range in which pH must fall to meet state standards. To meet state standards, dissolved oxygen must fall above red line in (b) and temperature must fall below red line in (c) (Colorado Department of Public Health and Environment, Water Quality Control Commission n.d.). Created using data from the SCPN (Southern Colorado Plateau Network 2024).

NPS / CONSERVATION SCIENCE PARTNERS

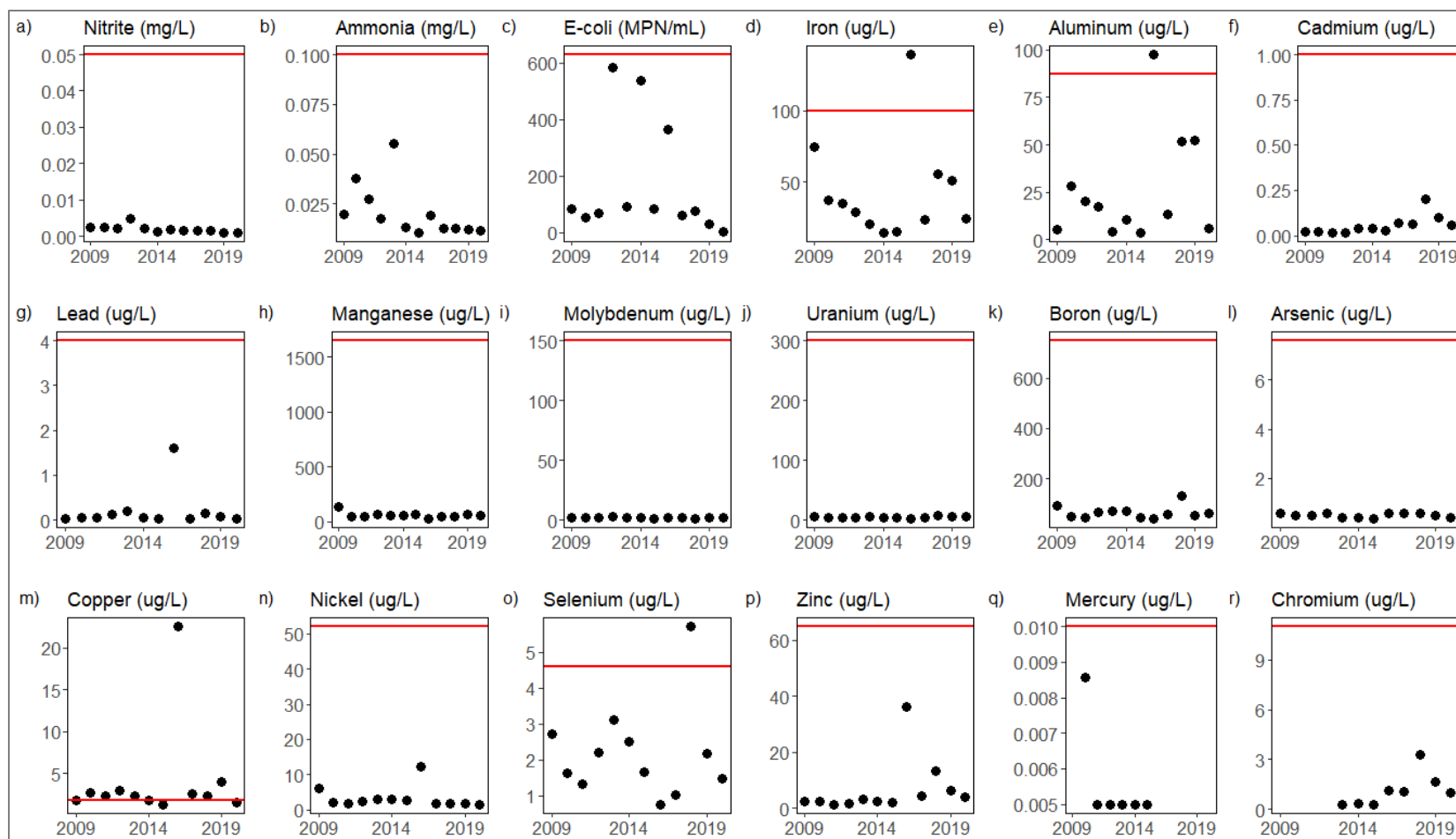


Figure 27. Average annual values of a) Nitrite (mg/L), b) Ammonia (mg/L), c) E. coli (MPN/mL), d) Iron (ug/L), e) Aluminum (ug/L), f) Cadmium (ug/L), g) Lead (ug/L), h) Manganese (ug/L), i) Molybdenum (ug/L), j) Uranium (ug/L), k) Boron (ug/L), l) Arsenic (ug/L), m) Copper (ug/L), n) Nickel (ug/L), o) Selenium (ug/L), p) Zinc (ug/L), q) Mercury (ug/L), and r) Chromium (ug/L) from 2009 to 2021. Some survey values were reported with a less-than symbol, and the rest were not. To address this inconsistency, we averaged all reported values, including those with the less-than symbol (a conservative approach, since values with less-than symbols were very small). Red lines reflect the state standards of each parameter, under which each trace metal or mineral must fall to meet state standards (Colorado Department of Public Health and Environment, Water Quality Control Commission n.d.). Created using data from the SCPN (Southern Colorado Plateau Network 2024).

NPS / CONSERVATION SCIENCE PARTNERS

While multiple water quality indicators exceeded standards occasionally, only Copper exceeded the state standard for multiple years (Figure 27m). Exceedances of the copper standard are not unexpected given there is evidence of a combination of both natural sources and upstream Legacy mining effects on the East Mancos River, where exceedances have been documented (Brown and Roberts 2021).

To complete the condition assessment for Mancos River water quality, we assessed the current condition status of indicators of inorganic/non-minerals, nutrients, and trace metals and minerals that meet state standards for water quality using the condition rating thresholds developed in Appendix A.3. Table 6 below illustrates the outcome of this exercise, based on the SCPN monitoring data.

Table 6. Water quality condition ratings summary, based on SCPN monitoring from 2009 to 2021. Condition ratings below are based on the Condition Rating Table in Appendix A.

Indicator	Condition	Trend	Condition Evaluation
Inorganic/non-mineral WQ indicators	Good	N/A	The condition is good because all indicators met/exceeded state standards from 2009 to 2021.
Nutrient indicators	Good	N/A	The condition is good because all indicators met/exceeded state standards from 2009 to 2021.
Trace metal and mineral indicators	Fair	N/A	The condition is fair because ~73% of indicators met/exceeded state standards from 2009 to 2021.

Condition Assessment Summary for Macroinvertebrates and their habitat

The Southern Colorado Plateau Inventory and Monitoring Network (SCPN) and other NPS park units, through the Vital Signs program, have identified aquatic macroinvertebrates as a high-priority Vital Sign for monitoring the ecological integrity of streams in parks (Thomas et al. 2006). As summarized in Stumpf (2024), the condition of the aquatic macroinvertebrate community is an important indicator of the integrity and health of freshwater ecosystems. This is because their relatively short life spans and habitat specificity make them good “barometers” of conditions in the stream system (e.g., Brasher et al. 2011; Graeber et al. 2017). As such, tracking the status and trends of the aquatic macroinvertebrate community can alert managers to stresses occurring within freshwater ecosystems—ranging from pollutants to streamflow reductions to thermal fluctuations (Stumpf 2024). We note that this is a different but complementary approach to discrete water samples described above. The use of biological monitoring data can provide an integrated assessment of waterbody health that extends beyond discrete water chemistry samples.

From 2008 through 2019, SCPN monitored the status of, and long-term trends in, the aquatic macroinvertebrate community and its habitat in the Mancos River within MEVE. Data on key community and physical habitat metrics were collected from two long-term monitoring sites on the river: the upstream (US) site, MAN01 at Anita’s Flat near the northern park boundary, and the downstream (DS) site, MAN02 near the downstream boundary of the park (see Stumpf 2024) for map showing the monitoring sites).

Results of this long-term monitoring, recently reported in Stumpf (2024), suggest that both monitoring sites experienced periods of thermal and hydrologic stress during the monitoring period. These stresses occurred concurrently, and synergistically affected many aspects of the macroinvertebrate community. Below we relay the most important findings of Stumpf (2024) regarding the current condition of the Macroinvertebrate community on the Park’s stretch of the Mancos River.

Flow trend. Stream discharge or flow was measured at both monitoring stations each time data was collected from 2008 to 2019. Mean discharge at the US site was $0.255 \text{ m}^3/\text{sec}$ ($n=10$) and $0.276 \text{ m}^3/\text{sec}$ at the DS site ($n=7$). To determine trends in flow over time Stumpf (2024) used the Mann-Kendall test (Kendall 1975; Mann 1945), which is a rank correlation test that produces a value, tau (τ), that is used to determine whether the metric reliably increases or decreases through time. Tau values, scaled from -1 to 1 , describe the strength of trend, where values approaching 1 indicate a positive trend, and values approaching -1 indicate a negative trend. In order to test for significant trends in flow, Stumpf (2024) used the Theil-Sen estimator, which is a non-parametric regression analysis that computes intercepts and slopes for each pair of samples in the dataset. While there was no observable trend in flow at the downstream monitoring site, Stumpf (2024) found a negative (decreasing) trend in flow velocity at the US monitoring site for the 2008–2019 period (Figure 28).

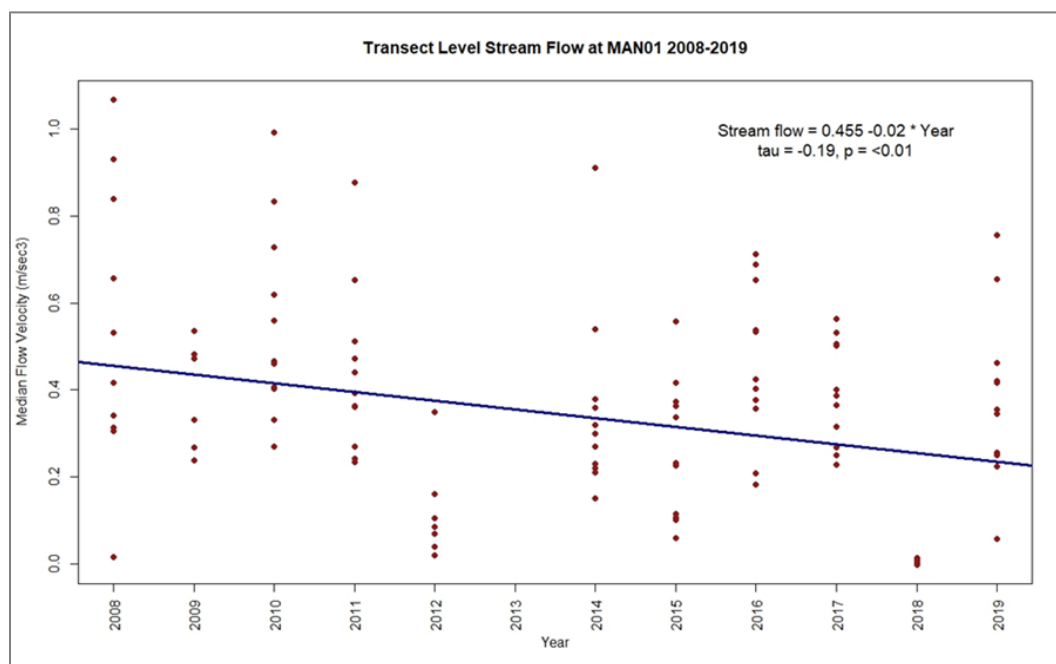


Figure 28. Streamflow trend over time at physical habitat transects at US Mancos SCPN monitoring site. Points represent median stream flow for each physical habitat transect during each monitoring year. The blue line represents the Theil-Sen estimated slope (which represents the median intercept and slope for all pairs of data) for negative trend in flow velocity at the US site. Figure produced from Stumpf (2024).

NPS / CONSERVATION SCIENCE PARTNERS

Tolerant vs. Intolerant Macroinvertebrate Taxa

Stumpf (2024) assessed taxonomic richness and diversity of benthic macroinvertebrates from samples collected from the Mancos River. Summaries of richness and diversity were completed at the genus level using a fixed-coverage rarefaction method. This method is preferred over summaries using raw counts because it can account for differences in annual sampling area, number of individuals collected, and laboratory subsampling differences across years, thus giving rare species similar weighting to common species. Stumpf (2024) determined 2008–2019 trends for pollution tolerant and intolerant taxa with the Mann-Kendall test and the Theil-Sen estimator (methods described above). While no observable trend was seen at the downstream monitoring site, at the US monitoring site Stumpf (2024) found that relative abundance of moderately tolerant taxa increased by an average of 2.60% a year ($\tau = 0.27$, $p = < 0.01$) (Figure 29 [top]), and the relative abundance of intolerant taxa decreased by an average 2.72% a year ($\tau = -0.26$, $p = < 0.01$) (Figure 29 [bottom]). Stumpf (2024) determined that these significant negative trends in abundance of intolerant taxa were likely caused by thermal and hydrologic stress, presumably due to stream warming and low flows.

Trichoptera trend. Caddisflies (order Trichoptera) are among the most sensitive groups of macroinvertebrates to swings in stream temperature (Houghton and DeWalt 2023; Stacey et al. 2021). For Trichoptera, Stumpf (2024) assessed richness using similar methods as those to assess richness of tolerant and intolerant taxa discussed above. Stumpf (2024) determined 2008–2019 trends for trichopteran taxa using methods described above (the Mann-Kendall test and the Theil-Sen estimator). From 2008–2019, Stumpf (2024) found that relative abundance of trichopteran taxa at the US site decreased by an average of -2.66% a year ($\tau = -0.26$, $p = 0.01$) (Figure 30 [top]), and trichopteran species richness decreased by an average of -0.64% a year ($\tau = -0.24$, $p = 0.02$) (Figure 30 [bottom]). Stumpf (2024) determined that the significant negative trends in abundance and richness were likely attributed to thermal and hydrologic stress.

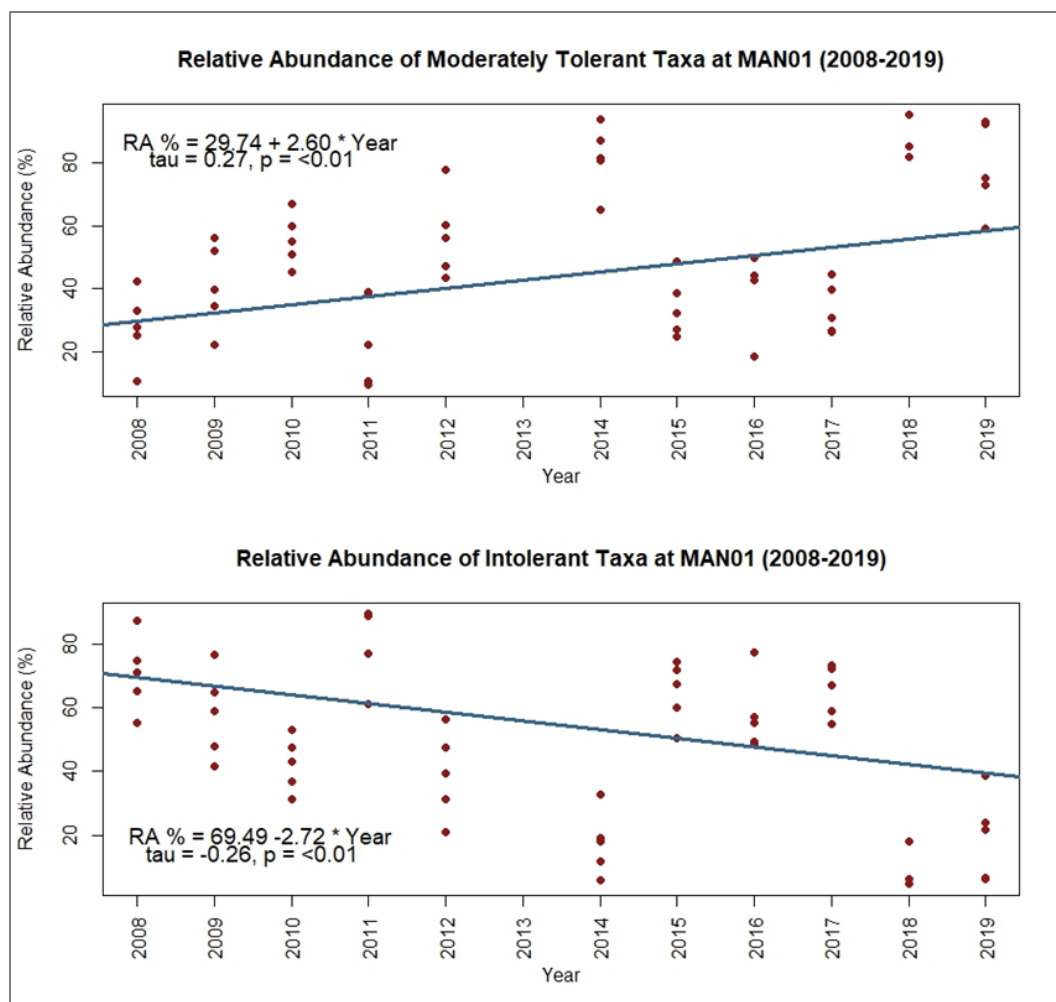


Figure 29. Trend in relative abundance of moderately tolerant (top) and intolerant taxa (bottom) at the SCPN upstream (US) monitoring site. The blue line represents the Theil-Sen estimated slope for trend in relative abundance. Figure produced from Stumpf (2024).

NPS / CONSERVATION SCIENCE PARTNERS

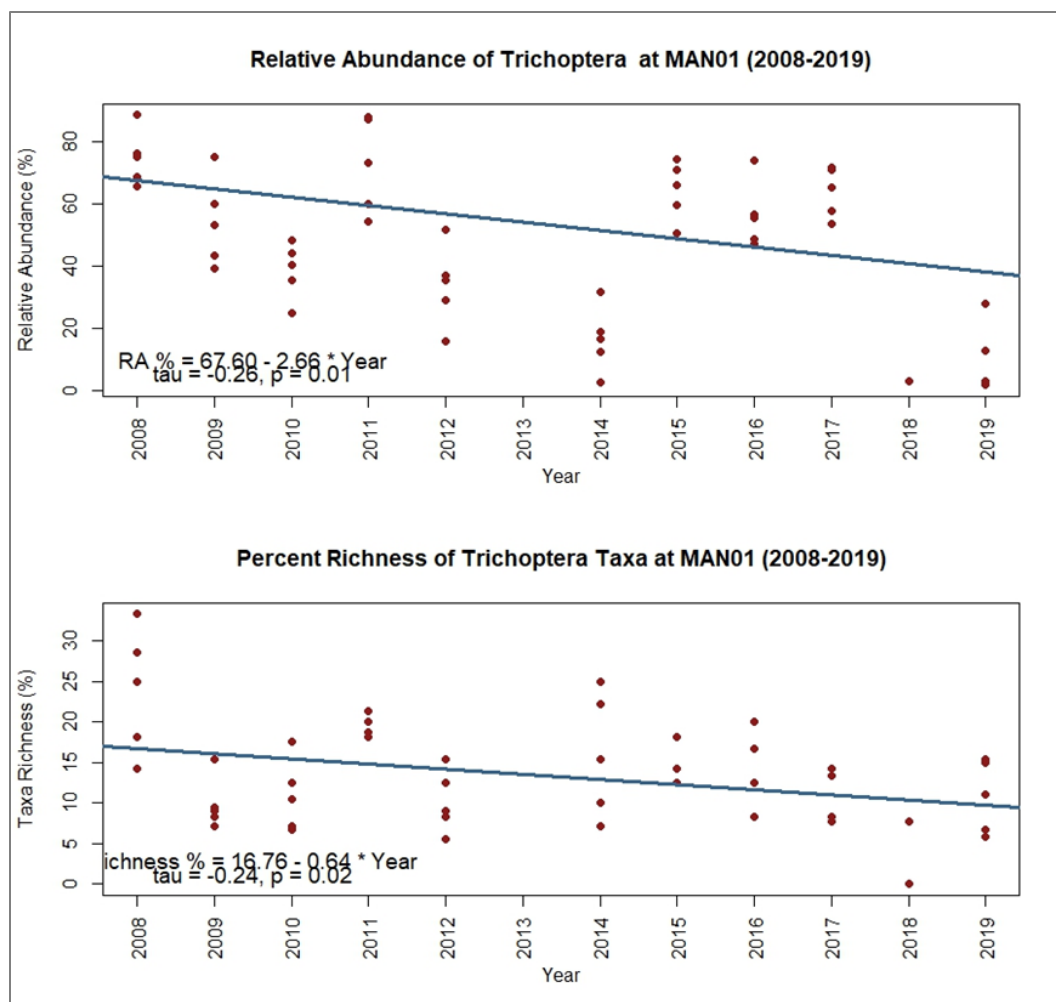


Figure 30. Trend in relative abundance (top) and species richness (bottom) of Trichopterans at the upstream SCPN monitoring site. The blue line represents the Theil-Sen estimated slope for trend in relative abundance. Figure produced from Stumpf (2024).

NPS / CONSERVATION SCIENCE PARTNERS

Percent of Samples that Meet Colorado's Water Quality Attainment Standards

In order to assess the ecological integrity of the macroinvertebrate community in MEVE's stretch of the Mancos River, Stumpf (2024) applied Colorado's Multi-Metric Index (MMI) Bioassessment Tool (Colorado Department of Public Health and Environment 2003) to the 2008–2019 sample data. The MMI determines whether the aquatic life of streams and rivers (in this case a warm-water fishery) meets attainment of the State's water quality standard, or whether the site is impaired. Metrics incorporated in the MMI include community richness, composition, and tolerance to disturbance, which can reflect a wide range of ecological stresses that can affect community attributes of structure, function, habit and sensitivity to disturbance (Colorado Department of Public Health and Environment 2003). Overall scores for the MMI are derived by combining scores for all of the individual samples. Combined MMI scores of less than 34 receive a designation of impaired, and combined samples scoring 45 or higher support a designation of attainment (scores between 34 and

45 are “gray areas” and have two additional metrics applied to them before an impairment or attainment designation can be made (Colorado Department of Public Health and Environment 2003).

Using the MMI tool on the 2008–2019 Mancos data, Stumpf (2024) reported that 28.3% of the upstream site samples and 20.5% of the downstream site samples did not meet the threshold for attainment of Colorado’s water quality standard for warm-water life. At the downstream site, the largest concentration of impaired samples occurred in 2015, when all samples were designated as impaired. Stumpf (2024) also noted that in both 2016 and 2018 samples were not collected at the downstream site due to drying conditions and a lack of flowing water at the monitoring site, and for those years the site should be considered in a state of impairment. Stumpf (2024) reported that there were five metrics that contributed most to impairment scores: (1) percentage of intolerant taxa, which showed the greatest decline, decreasing by 68.13% over the sample period, (2) the number of EPT² taxa, which decreased by 35.47%, (3) abundance of facultative predators and (4) “shredders” (that break down organic materials), which together decreased by 52.40%, and (5) abundance of “clinger taxa” (that attach themselves to the underside of substrates), which decreased by 38.91% over the sample period.

To complete the condition assessment for aquatic macroinvertebrates, the current condition status of indicators of flow trend, relative abundance of tolerant versus intolerant taxa, Trichoptera trends, and percent samples that meet State standards for water quality impairment were assessed using the condition rating thresholds developed in Appendix A.4. Table 7 below illustrates the outcome of this exercise, based on the SCPN monitoring data and analyses for macroinvertebrates provided by Stumpf (2024).

Table 7. Macroinvertebrate condition ratings summary, based on SCPN monitoring from 2008 to 2019. Condition ratings below are based on the Condition Rating Table in Appendix A.

Indicator	Condition	Trend	Condition Evaluation
Flow trend	Fair/Poor	Downward	The condition is fair/poor because of a significant negative trend in flow velocity at the US sample site, while there was also a (non-significant) downward trend for flow velocity at the DS site.
Relative abundance of pollution tolerant vs. intolerant taxa	Poor	Downward for intolerant taxa, upward for tolerant taxa	The condition is poor because relative abundances of moderately tolerant taxa significantly increased, while relative abundances of intolerant taxa significantly decreased.
Trichoptera trend	Poor	Downward	The condition is poor because Trichoptera experienced significant negative trends in abundance (US monitoring site).
Percentage of samples that meet CO threshold for attainment	Fair/Poor (US sample site)	N/A	The condition at the US site is fair/poor because 71.7% of samples met the Colorado State standard for attainment for warm-water aquatic life.

² Ephemeroptera (mayflies), Plecoptera (stoneflies), and Trichoptera (caddisflies).

Table 7 (continued). Macroinvertebrate condition ratings summary, based on SCPN monitoring from 2008 to 2019. Condition ratings below are based on the Condition Rating Table in Appendix A.

Indicator	Condition	Trend	Condition Evaluation
Percentage of samples that meet CO threshold for attainment (continued)	Fair (DS sample site)	N/A	The condition at the Downstream site is fair because 79.5% of samples met the Colorado State standard for attainment for warm-water aquatic life.

Condition Assessment Summary for Fishes and Aquatic Habitat

Fish Counts and Abundances Over Time

We acquired the long-term fish survey dataset from Colorado Parks and Wildlife (CPW) for the Mancos River, for 1944–2022 (Colorado Parks and Wildlife 2023). This is a comprehensive dataset of all known fish surveys on the Mancos River and includes the numbers of fish per species caught and the relative abundance of species per survey effort (i.e., individuals of a species caught/total individuals caught). We used relative abundances of species for analyses, rather than total catch per species, to ameliorate very large swings in total catch in instances of very low catch rates (i.e., perhaps 10 individuals total per survey) during very low flows. In instances where a fish species was documented at multiple surveys in a single year, multiple relative abundances were recorded for that species within the dataset. We also added to the CPW dataset a 2022 National Park Service fish survey of the Mancos River within MEVE (National Park Service 2024) that captured similar metrics. We cross-checked that no surveys were missing from this combined dataset using a compendium of fish surveys conducted on San Juan River tributaries from the 1960s to 2000 (Miller and Rees 2000). While detailed information on specific survey sites was missing from Miller & Rees (2000), most surveys included in Miller and Rees (2000) for the Mancos River were also included in the combined dataset.

In order to favor capturing fishes typically found in a warm-water fishery, and to control for possible high spatial variability in fish community composition due to hydrology, water quality, geomorphology and water management, survey sites used in our analyses were stratified by elevation: only Mancos River locations lower than 7,000 ft, the approximate elevation of the town of Mancos, were selected for the temporal analyses of fish community composition within MEVE (personal communication, Nic Medley, NPS—Water Resource Division, January 2024). Even though total fish catch levels in some years with low flow levels (in some instances near dry) were quite low, relative abundances of species over the long-term dataset was determined to be a helpful metric to analyze, given the large swings in overall fish numbers depending on flow levels.

We analyzed the relative abundance of native and non-native fishes from 1944 to 2022 using Generalized Least Squares (GLS) regression models. All models accounted for autocorrelation between yearly observations. Data were log-transformed and GLS regression models were applied to native and non-native relative abundances separately. There was not a significant effect of year on the relative abundance of the native guild ($\beta_1 = 0.0008$, $p\text{-value} = 0.8699$), suggesting that collectively this guild was relatively stable over time (Figure 31). For the non-native guild, there was

a statistically significant negative slope ($\beta_1 = -0.0315$, $p\text{-value} = 1e-04$), indicating a downwards trend of this guild (Figure 31). Assumptions were tested and residuals were normally distributed via a Shapiro test ($p\text{-value} > 0.05$) and no significant autocorrelation was detected (Ljung-Box test $p\text{-value} > 0.05$). Further details on this methodology can be found in Appendix B.5.

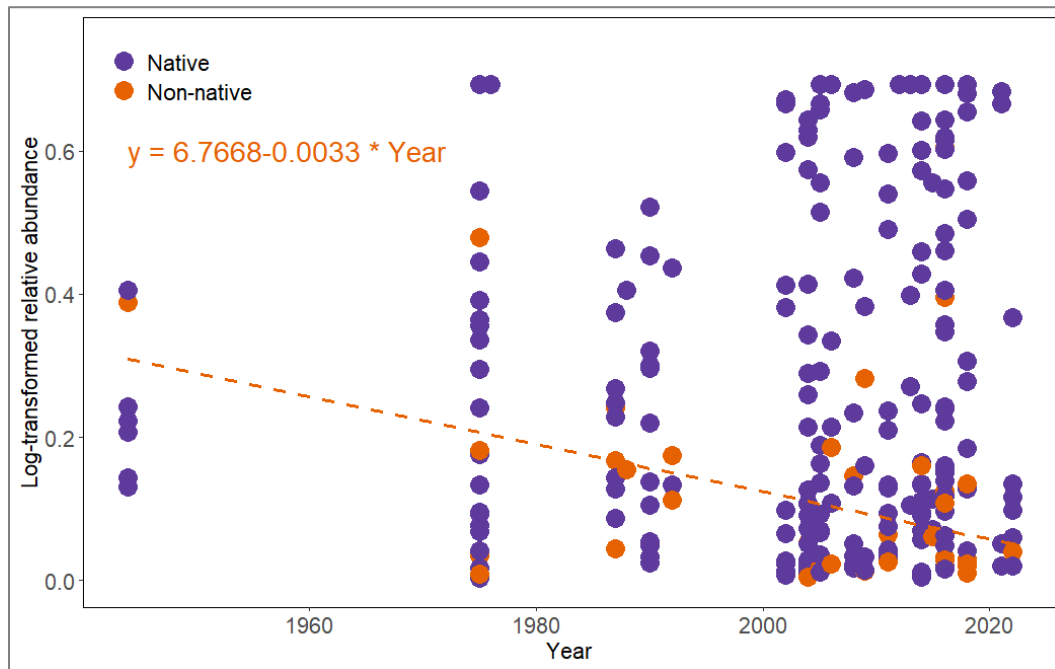


Figure 31. Log-transformed relative abundance (individuals caught of species / total individuals caught) estimates for each individual species from 1944 to 2022. Relative abundance estimates are color-coded based on if they represent a native or non-native fish. The GLS model's fitted values from the non-native guild are represented as a trendline. The associated equation of the trendline is also reported. The GLS model of the native guild was not reported because there was not a significant effect of year on the relative abundance of the native guild. Created using data from Colorado Parks and Wildlife (2023).

NPS / CONSERVATION SCIENCE PARTNERS

Experts with CPW and NPS expressed concern that the speckled dace (*Rhinichthys osculus*) skewed the data towards a healthier population of native fish than actually exists. To address this concern, we analyzed the relative abundance of the native guild without speckled dace from 1944 to 2022 utilizing the same GLS regression model as in Figure 32. There was no significant effect of year on the relative abundance of the native guild without the speckled dace ($\beta_1 = -0.0016$, $p\text{-value} = 0.2031$), reconfirming the relative stability of the native guild during this time period.

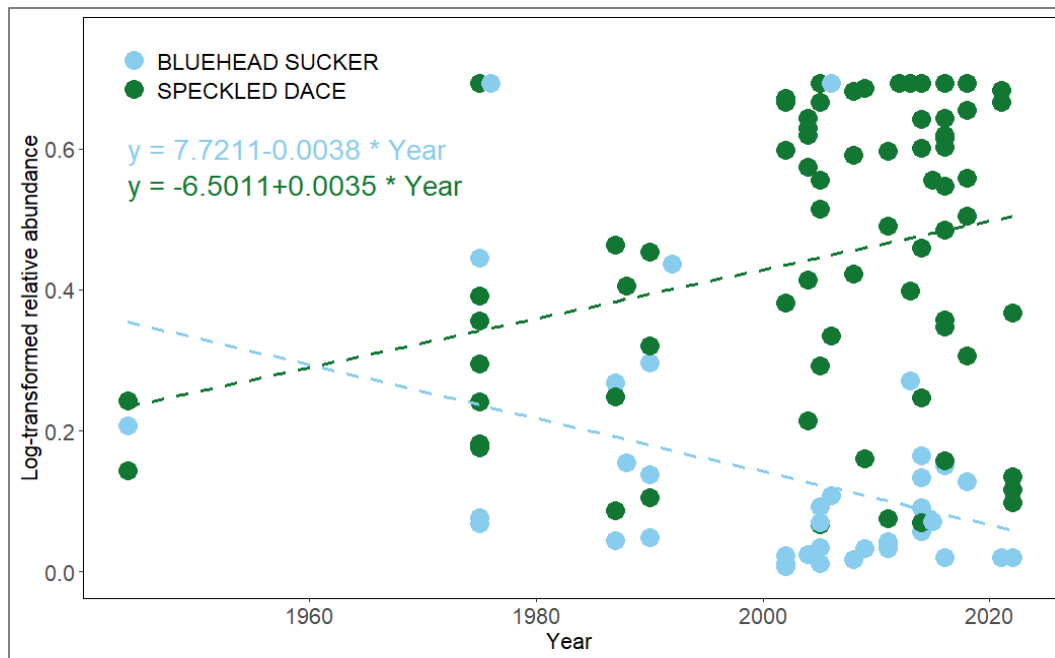


Figure 32. Log-transformed relative abundance (individuals caught of species / total individuals caught) of bluehead sucker and speckled dace from 1944 to 2022. The GLS model's fitted values from both species are represented as trendlines and the associated equations of the trendlines are reported. The relative abundances and GLS models of all other species in the dataset were not reported because there was not a significant effect of year on the relative abundance of all other species. Created using data from Colorado Parks And Wildlife (2023).

NPS / CONSERVATION SCIENCE PARTNERS

We analyzed the relative abundance of all native and non-native species individually from 1944 to 2022 using GLS regression models. Data were log-transformed and GLS regression models were applied to each species' relative abundances separately. There was a significant effect of year on the relative abundance of the bluehead sucker [*Catostomus discobolus*] ($\beta_1 = -0.0038$, p-value = 0.0251) which declined³, and for speckled dace ($\beta_1 = 0.0035$, p-value = 0.0159), which increased over time (Figure 32). Further details on this methodology can be found in Appendix B.5. There was no significant effect of year on the relative abundances of all other species (black bullhead [*Ameiurus melas*], bluegill sunfish [*Lepomis macrochirus*], brown trout [*Salmo trutta*], channel catfish [*Ictalurus punctatus*], common carp [*Cyprinus carpio*], fathead minnow [*Pimephales promelas*], flannelmouth sucker [*Catostomus latipinnis*], green sunfish [*Lepomis cyanellus*], largemouth bass [*Micropterus salmoides*], mottled sculpin [*Cottus bairdii*], rainbow trout [*Oncorhynchus mykiss*], red shiner [*Cyprinella lutrensis*], Rio Grande chub [*Gila pandora*], roundtail chub [*Gila robusta*], sand shiner [*Notropis stramineus*], and yellow perch [*Perca flavescens*]; all p-values > 0.05).

Figure 32 indicates that speckled dace have steadily increased in terms of relative abundance in the Mancos River. Speckled dace are highly adaptable to drying conditions; for example, this is typically

³ The 2000 Bircher fire likely also contributed to declining numbers of some species of fish, due to post-fire sedimentation into the river (Coyler 2006).

the first species to return after river segments dry up (personal communication, Nic Medley, NPS-Water Resources Division, October 2024). As such, recent relative abundances of speckled dace on the Mancos River are not representative of the larger fish community in many other western warm-water native streams and rivers (personal communication, Jim White, Colorado Parks and Wildlife, September 2024). Also of note is that roundtail chub relative abundances did not significantly increase in our analyses despite being stocked intermittently from 2003 to 2016, itself a matter of concern for this native species receiving special management attention.

Fish Habitat and River Flows

Figures 26 and 27, and the final Condition Ratings in the Water Quality section above, illustrate that water quality, when water is present, is currently considered good (i.e., it mostly meets CO water quality standards) in the Park's stretch of the Mancos River. Thirteen of the 21 water quality parameters assessed to reach this conclusion have state water quality standards that are specifically for aquatic life (Colorado Department of Public Health and Environment, Water Quality Control Commission n.d.). Thus, a good water quality condition rating for the Mancos River in the Park confers good fish habitat as well (assuming flows continue through the summer months, which is not always the case), and acts in concert with physical conditions that supports spawning and good macroinvertebrate habitat, a primary food source.

We also considered fish habitat quality in relation to the most recent results generated from the application of the Rapid Stream Riparian Assessment (RSRA) protocol (Stacey et al. 2021) within MEVE's stretch of river, which occurred in July of 2024. The RSRA protocol scores field indicators with a 1–5 point system in five functional categories: Water Quality, Fish and Aquatic Habitat, Fluvial Geomorphology, Riparian Vegetation, and Riparian Structure and Wildlife Habitat. For this portion of the Condition Assessment for Mancos River fishes we used the Fish and Aquatic category of the RSRA survey.

The most recent (July 2024) results of the RSRA survey on the MEVE stretch of the Mancos River⁴, performed near the downstream boundary with the Ute Mountain Ute Reservation, generated a score of 3.0 for the Fish/Aquatic Habitat Category. While pool/riffle distribution with this RSRA Study reach scored as excellent (5.0), this was offset by average scores (3.0) for macroinvertebrate diversity, presence of large woody debris, and overbank cover. Underbank cover (2.0) and cobble embeddedness (2.0) scored poorly. Based on these scores, we considered fish/aquatic habitat on the Mancos River to be Fair, and stable, because prior RSRA surveys performed on the same stretch in the Park scored similarly: 2.5 (2006) and 3.5 (2012)⁵.

We performed an independent assessment of Mancos River discharge that was fish focused. We analyzed data from the closest downstream gage from MEVE on the Mancos River, the Towaoc

⁴ 2024 RSRA score sheet and data forms are in MEVE files. Andrew Spear, MEVE NP Physical Science Program Manager, provided these to CSP in August 2024.

⁵ While RSRA surveys were performed by the Mancos Watershed Group and Mancos Conservation District both upstream and downstream of MEVE in 2018, we were not able to use this datapoint in assessment of trend because the surveys were not done in the same stretch of the Mancos as the 2006, 2012 and 2024 surveys.

Gage (USGS Gage# 09371000), on the Ute Mountain Ute Reservation. We did not analyze flow trends from the closest upstream gage to the Park at Anita's Flat (USGS Gage # 09370600), because this gage was only utilized from 2003 to 2015. Despite the Towaoc Gage being impacted by runoff events lower in the watershed than the stretch of the mainstem located in MEVE, the gage provides some information on the Mancos River hydrograph for a span of over 100 years (1921–2024). Daily minimum and maximum flow summaries have been available since 1975. We analyzed minimum flow (cubic feet per second) (Figure 33a) and maximum flow (cubic feet per second) (Figure 33b) per year at the Towaoc Gage via Generalized Least Squares regression models. Data were log-transformed and GLS regression models were applied to minimum and maximum flows separately. Results suggest a negative trend in minimum flows ($\beta_1 = -0.0239$, p-value = 0.0020) (Figure 33a) and maximum flows ($\beta_1 = -0.0210$, p-value = 0.0194) (Figure 33b). Assumptions were tested via a plot of residuals vs fitted values, Q-Q plots for normality and no significant autocorrelation was detected (Ljung-Box test p-value > 0.05). Further details on this methodology can be found in Appendix B.6. Of potential concern, especially for the fish community, is that minimum flows are increasingly approaching zero each year, which happened only intermittently in the past. Further, it appears that maximum flows are becoming much more variable (with higher peaks and lower lows).

We analyzed “same day” river flow during fish surveys compared to relative abundance of natives and non-natives from 1975 to 2022 (river flow data was unavailable in 1944), but no significant trends were found. Declines in flows coincided with a decline in non-native fish relative abundances while native fish relative abundances remained stable (see Figure 31). While we cannot determine this from our data, it is possible that some warm-water native fish species (which presumably evolved with drying periods over millennia), might fare better with increased instances of drying (i.e., seeking temporary refuge in disconnected pools) than non-native species.

To complete the condition assessment for fishes, the current condition status of indicators of fish community diversity, fish habitat quality, and river flow over time, were assessed using the condition rating thresholds developed in Appendix A.5. Table 8 below illustrates the outcome of this exercise. We have a medium confidence level in the condition rating results for fish community diversity because often the catch rates were very low when flows were very low; we have a high confidence level in the condition rating results of the other indicators. However, we were reluctant to assign a trend for fish community diversity, as fish survey results are so tied to whether the river is flowing through the Park at all, and all indications are that drying spells within MEVE are going to continue, or even become more frequent with climate change.

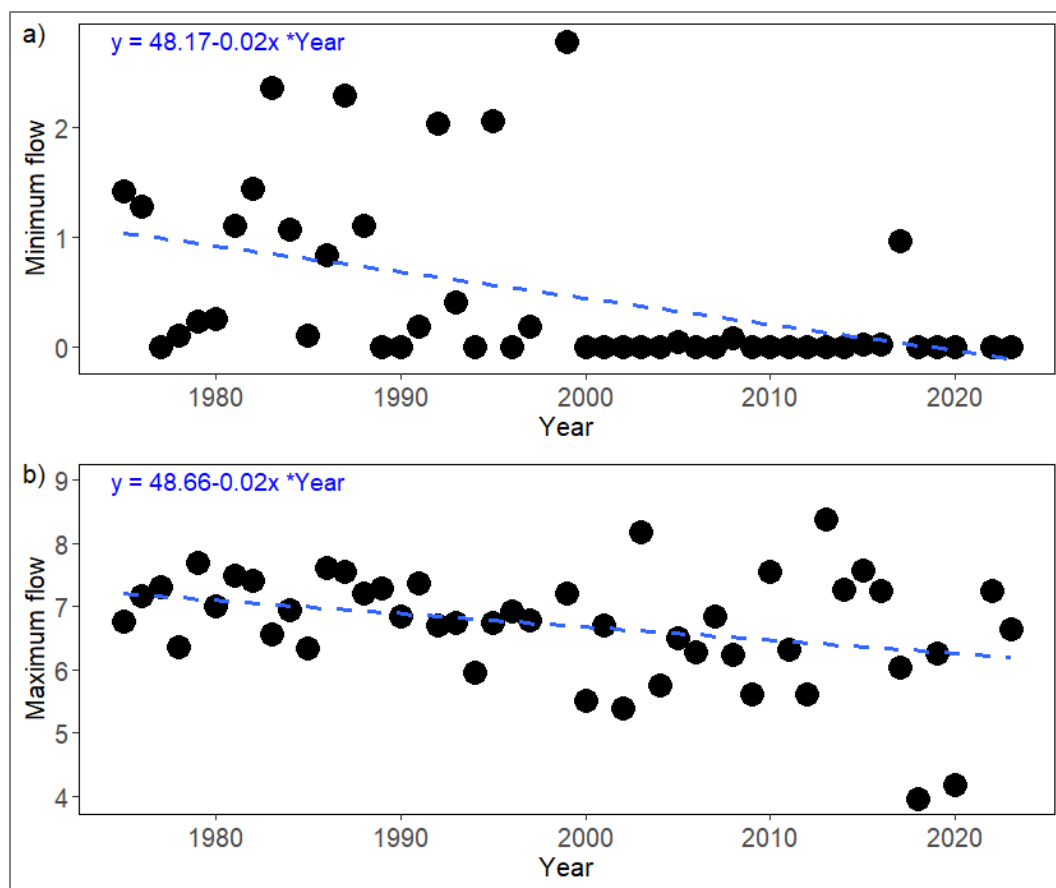


Figure 33. Log-transformed a) minimum flow (cubic feet per second) and b) maximum flow (cubic feet per second) reported each year at the Towaoc Gage (USGS Gage# 09371000) from 1975 to 2022. The fitted values from both GLS models were represented as trendlines with the associated equations reported. Created using data from the Towaoc Gage (USGS GAGE# 09371000).

NPS / CONSERVATION SCIENCE PARTNERS

Table 8. Mancos River fish and aquatic habitat condition ratings summary. Condition ratings below are based on the Condition Rating Table in Appendix A.

Indicator	Condition	Trend	Condition Evaluation
Current Fish Community Diversity	Fair	N/A	The condition is fair because even though relative abundances of native fish are greater than non-natives, it is unbalanced in terms of relative abundances of natives.
Habitat Quality (using Fish/Aquatic Indicators from RSRA surveys)	Fair	Stable	The condition is fair because the average score for the Fish/Aquatic habitat category from the most recent (2024) RSRA Survey was 3.0.
Flow (using closest stream gauge for longest period of record)	Poor	Downward	The condition is poor because linear regression models performed on minimum and maximum flows from 1975 to 2022 revealed significant negative trends.

Condition Assessment Summary for Riparian and River Corridor

For the condition assessment of the Mancos River's riparian and river corridor, we primarily focused on the most recent results generated from the application of the Rapid Stream Riparian Assessment (RSRA) protocol (Stacey et al. 2021) within MEVE's river segment. However, for this part of the Condition Assessment, we only focused on the RSRA's last three functional categories. This is because we have a more robust assessment of water quality supplied by SCPN (discussed above), and the Fish and Aquatic category was utilized above in the Fish Condition Assessment.

The RSRA protocol was used to assess the health and function of the Mancos River through MEVE in 2006 (Stacey 2007), 2012, and 2024. This allowed for a general assessment of trend for the function of the river and riparian system over this time period⁶. The same assessment reach, near the downstream boundary of the Park, was used for all three surveys. The survey reach was initially selected in 2006 because the team deemed it a representative reach of the Mancos River through MEVE, and thus similar to the habitat available through the corridor that spans MEVE managed lands (Stacey 2007).

The July 2024 RSRA within the Park's stretch of the river was conducted by MEVE Park staff to determine whether any of the functional categorical scores had shifted since the 2012 survey. With average scores for riparian vegetation health and function of 4.0 in 2006, 4.1 in 2012, and 4.1 again in 2024, this component of the riparian system is currently in good condition, with comparable RSRA scores across the three survey years (Table 9). The high degree of health and functionality of the riparian forest through MEVE is partly attributed to diverse demography (all age classes present) of both shrub and tree components of the riparian zone, as well as near absence of non-native woody species, and very light grazing and browsing pressure. Similarly, the trend in riparian structure and wildlife habitat over time is stable, although the current condition of this component of the riparian forest on MEVE's stretch of the Mancos River is Fair (Table 9). Field indicators that have consistently contributed to this rating (in 2006, 2012 and 2014) include poor to fair scores (2s and 3s) for both mid and upper canopy density of cottonwood trees (both *Populus deltoides* and *P. angustifolia*).

When comparing the RSRA scores over time, the 2024 average fluvial geomorphology score (3.4) indicates an upward trend, especially when compared to the 2006 score for this category (2.4). Two parameters contributed to improvement of this score over time: floodplain connectivity (score of 3.0 in 2024 compared to scores of 1.0 in both 2006 and 2012), and presence and activity of beaver (no presence, or a score of 1.0, in 2006, and recent activity but no dams, or a score of 3.0, in 2012). The presence of actively maintained beaver dams in 2024 raised this score for fluvial geomorphology to a 4.0. MEVE staff have been well aware of historic channelization in the Park's stretch of the river (Colyer 2006; Whitenack 2016), and one of the goals of the Park's river restoration efforts has been to reconnect the river channel with the historic floodplain. According to the 2024 RSRA survey

⁶ The Mancos Watershed Group (MWG) and the Mancos Water Conservancy District were partners with MEVE for the 2006 and 2012 surveys. The MWG and the Conservancy District began conducting annual RSRA surveys up and down the Mancos starting in 2018. The 2024 RSRA survey for the MEVE stretch of the Mancos is the most recent survey in the Park boundaries since 2012, and can be added to this now long-term RSRA dataset for the river.

results, this improvement could be happening. Moreover, increased beaver activity, including dam building, is expected with the successful implementation of Post Assisted Log Structures (PALs) (Bennet et al. 2019). As we discuss in Chapter 4, a synergy should result on the MEVE river segment if implementation of PALs continue, and beavers move in and remain to keep up the structures, while continuing to augment the channel with dams of their own.

Table 9. Riparian and river corridor condition ratings summary. Condition ratings below are based on the Condition Rating Table in Appendix A.

Indicator	Condition	Trend	Condition Evaluation
Fluvial Geomorphology	Good/Fair	Stable to upwards	The condition is good/fair because the average score for the fluvial geomorphology category from the most recent (2024) RSRA Survey was 3.4 out of 5.
Riparian Vegetation	Good	Stable	The condition is good because the average score for the riparian vegetation category from the most recent (2024) RSRA Survey was 4.1 out of 5.
Riparian Structure & Wildlife Habitat	Fair	Stable	The condition is fair because the average score for the riparian structure and wildlife habitat category from the most recent (2024) RSRA Survey was 3.0 out of 5.

MEVE falls within the known current or historic range of the yellow-billed cuckoo (*Coccyzus americanus*), southwestern willow flycatcher (*Empidonax traillii extimus*) and New Mexico meadow jumping mouse (*Zapus hudsonius luteus*)⁷, all federally listed riparian wildlife species. While none of these riparian species are currently known within MEVE (personal communication, Nathan Brown, MEVE NP Physical Science Program Manager, October 2024), if riparian habitat within the Park continues to improve, perhaps these species could eventually move back in. The cuckoo is a migratory species in this part of Colorado and prefers a robust and well-connected upper canopy layer. All three (2006, 2012 and 2024) RSRA surveys indicate poor scores (2.0) for the upper riparian canopy parameter, indicating sub-par habitat for the cuckoo in MEVE. The southwestern willow flycatcher prefers a vigorous and connected shrub layer, preferably composed of willows (Sogge et al. 1997). All three prior RSRA surveys measured high shrub cover: 74% in 2006, 61% in 2012, and 80% in 2024, however the survey did not break out shrub cover by species (e.g., *Salix*) or patch size, which are both important metrics when determining willow flycatcher habitat suitability (US Fish and Wildlife Service 2002). The meadow jumping mouse prefers moist riparian meadow and riparian understory habitat (Zwank et al. 1997). While the ground cover layer of the riparian zone was shown in the two most recent RSRA surveys to be substantial (66% to 84% cover⁸), the very poor scores in all survey years for the extent of non-native herbaceous species in the ground cover layer (2, 1, and 1 for 2006, 2012 and 2024, respectively) may indicate only modest habitat benefits for the meadow jumping mouse in MEVE, especially since the RSRA survey does not break

⁷ However, Mesa Verde NP falls outside of the range of New Mexico meadow jumping mouse according to the current USFWS Information for Planning and Consultation System.

⁸ Raw cover data is not available for the 2006 RSRA survey.

out ground cover by species (e.g., *Carex*) or patch size, both important metrics when determining meadow jumping mouse habitat suitability.

To complete the condition assessment for riparian and river corridor condition, the current condition status of indicators of fluvial geomorphology, riparian vegetation, and riparian structure and wildlife habitat, were assessed using the condition rating thresholds developed in Appendix A.6. The rating system we developed for this exercise ties directly to the 5-point scoring system of the RSRA protocol. Table 9 below illustrates the outcome of this exercise. We note that due to the time scales upon which riparian restoration operates, one would expect to see riparian vegetation scores improve more slowly than fluvial geomorphology, and riparian structure and wildlife habitat even more slowly than vegetation.

3.5. Upland Vegetation System

The piñon-juniper (PJ) woodland of MEVE is of high conservation concern due to the high level of biodiversity it supports and ecosystem services it provides (Rondeau et al. 2011). In MEVE, PJ woodland consists of *Pinus edulis* (two-needle piñon pine) and *Juniperus osteosperma* (Utah juniper) and occurs on most mesas and upper canyon slopes up to 2,400 m elevation (Thomas et al. 2009). The Park supports remnant old growth stands of PJ woodlands that are 300–500 years old (Turner et al. 2007), with some individual trees aged over 500 years (Floyd et al. 2003). These contemporary woodlands also coincide with where the Ancestral Pueblo cleared fields for agriculture and built villages (Pueblo I and Pueblo II periods). The northern half and eastern portion of the Park hosts a patchy mosaic of woodlands and shrublands, mixed with species such as Gambel oak, mountain snowberry (*Symphoricarpos oreophilus*), and cliff fendlerbush (*Fendlera rupicola*) (Thomas et al. 2009).

Fire has played an important role in shaping the recent floristic ecology of MEVE, particularly within PJ habitat. Historically, persistent PJ woodlands in this part of the Colorado Plateau have been thought to sustain infrequent, high-severity crown fires with a return-interval of 400 years or more (Floyd et al. 2004; Turner et al. 2007). In general, regeneration of PJ woodlands tends to be slow and requires many decades for new trees to re-establish after stand-replacing fires (Floyd et al. 2000a). Currently, with the combined stressors of climate change and drought, a substantial amount of MEVE's PJ woodlands have burned in severe fires (Figure 34). In fact, two-thirds of the PJ woodlands in the Park have experienced fire from 1996 to 2006 (Turner et al. 2007). In many areas that have burned it is still too early to observe the recolonization of PJ woodlands (Erdman 1970), and currently these post-fire landscapes are dominated by fast-resprouting shrub species.

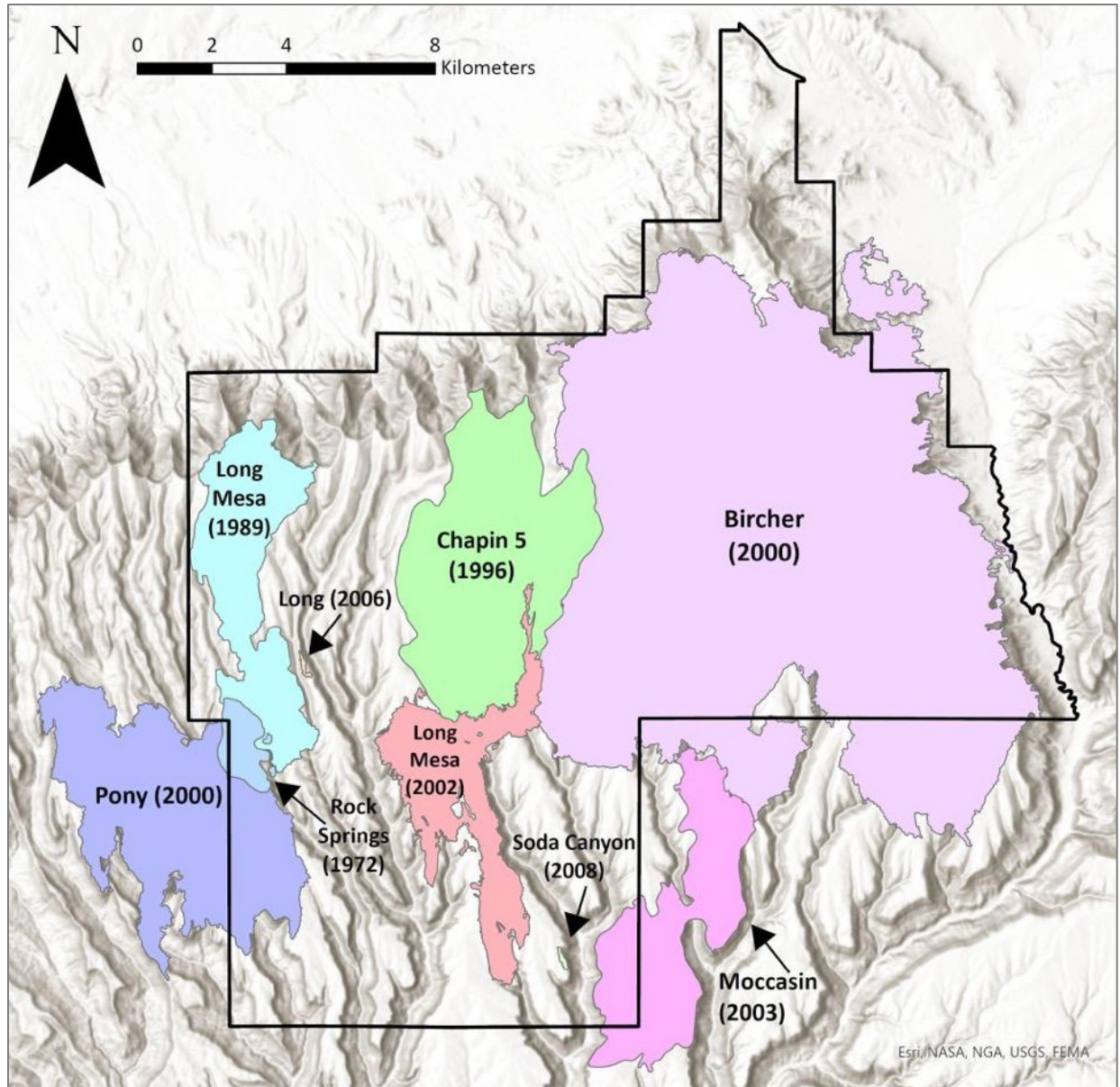


Figure 34. Recent fire history in MEVE. Fire perimeters shown are those within the park boundary and exceeding 10 acres in total from 1986 to 2023. Created using data from the National Interagency Fire Center (2022).

NPS / CONSERVATION SCIENCE PARTNERS

Mountain shrubland is another ecologically important vegetation type across MEVE, and it has become more prevalent in recent years (Appendix D). In MEVE this habitat is characterized by the prevalence of Gambel oak, although other shrubland species like Utah serviceberry (*Amelanchier utahensis*), fendlerbush, and mountain mahogany (*Cercocarpus montanus*) may co-dominate (Thomas et al. 2009). In MEVE, mountain shrubland occurs between elevations of about 2,100 and 2,400 m (6,750–8,000 ft) and where soils are deep and well-developed, consisting of sand, sandy loam, and sandy silt (Thomas et al. 2009). When disturbance events like fire result in the removal of

above-ground stems, oak and serviceberry assemblages are quick to resprout, demonstrating the fire-adapted nature of these species (Kaufmann et al. 2016).

MEVE includes several other, less common upland vegetation types that occupy a smaller portion of the Park. The shrub-steppe community is dominated by big sagebrush (*Artemisia tridentata*), which occurs at lower elevations in many of the canyon bottoms, in some areas where fire has removed PJ canopy, and in the mountain shrubland-PJ woodland ecotone (Thomas et al. 2009). Douglas fir (*Pseudotsuga menziesii*), ponderosa pine (*Pinus ponderosa*), and quaking aspen (*Populus tremuloides*) occur at higher elevation side canyon microsites in the Park, where high, narrow canyon walls provide shade and allow for the retention of snow, yielding moister conditions. Fires in the early 2000s have greatly reduced the proportion of Douglas fir and ponderosa pine stands throughout the Park, and these species are showing low levels of post-fire recruitment two decades later, likely due to warming temperatures (Thomas et al. 2009).

Increases of non-native plant species have been another relatively recent (compared to historic times) driver acting on this system. MEVE tracks 18 species⁹ of noxious weeds in the Park (Mesa Verde National Park 2022) and applies treatments to patches of noxious weeds each year. Treatment tactics are contained in the annual Invasive Plant Treatment Plans which are meant to build on the accomplishments from previous years.



Utah juniper (*Juniperus osteosperma*). With permission of Al Schneider.
AL SCHNEIDER

⁹ Not all of these are upland species; four of them principally occur in riparian zones.

Upland Vegetation Gap Analysis Summaries

Quantification of Vegetation Cover Change

To evaluate change in coarsely classified vegetation distribution across MEVE, we used the Rangeland Analysis Platform (Allred et al. 2021). This dataset is modeled using the long-term Landsat collection imagery record trained on 75,000 field plots of rangeland ecosystems and is produced annually at 30 m spatial resolution since 1984. We queried data for MEVE from the Google Earth Engine catalog, which includes years 1986–2023. RAP data consist of fractional cover for six categorical vegetation cover classes: bare ground, litter, annual forbs and grasses, perennial forbs and grasses, shrub, and tree cover. As these data are presented as continuous values (0–100% cover) rather than discrete categorical labels simplifying mixed-type land cover to a single cover type, RAP data provide an opportunity to evaluate subtle changes in vegetation composition over time. While RAP data are well-established and transferable beyond the perimeter of MEVE, they do not represent the vegetation of MEVE and vicinity at its most granular level (e.g., species diversity). However, this dataset is appropriate for a gap analysis because it reveals changes in broad categories of vegetation cover and can indicate transitions in successional state, including in response to disturbance events (e.g., fire) and long-term environmental pressures (e.g., climate).

At the Park-wide scale (i.e., aggregate area), analysis of RAP data revealed a general trend of declining tree cover and increasing shrub cover in MEVE over the period of analysis (Figures 35 and 36). Tree cover was dramatically reduced by the Bircher and Pony fires in 2000 and remained lower through the period of record with evidence of gradual recovery only in the last decade (Figure 36). Aggregate shrub cover showed the greatest variation across the period of record (coefficient of variation, $CV = 0.99$) and showed an overall increasing trend. Though annual forbs and grasses had the lowest aggregate cover in MEVE, this cover type showed a gradually increasing trend with moderate long-term variation ($CV = 0.39$). Perennial forbs and grasses also gradually increased over time but showed limited long-term variation ($CV = 0.16$). Similarly, bare ground ($CV = 0.32$) and litter ($CV = 0.13$) showed limited interannual variation. We also quantified decadal trends of change in fractional cover across MEVE and relative change in fractional cover within select areas affected by recent wildfires. More detailed explanation of RAP data, spatial and temporal analyses we conducted, and quantitative results can be found in Appendix E.

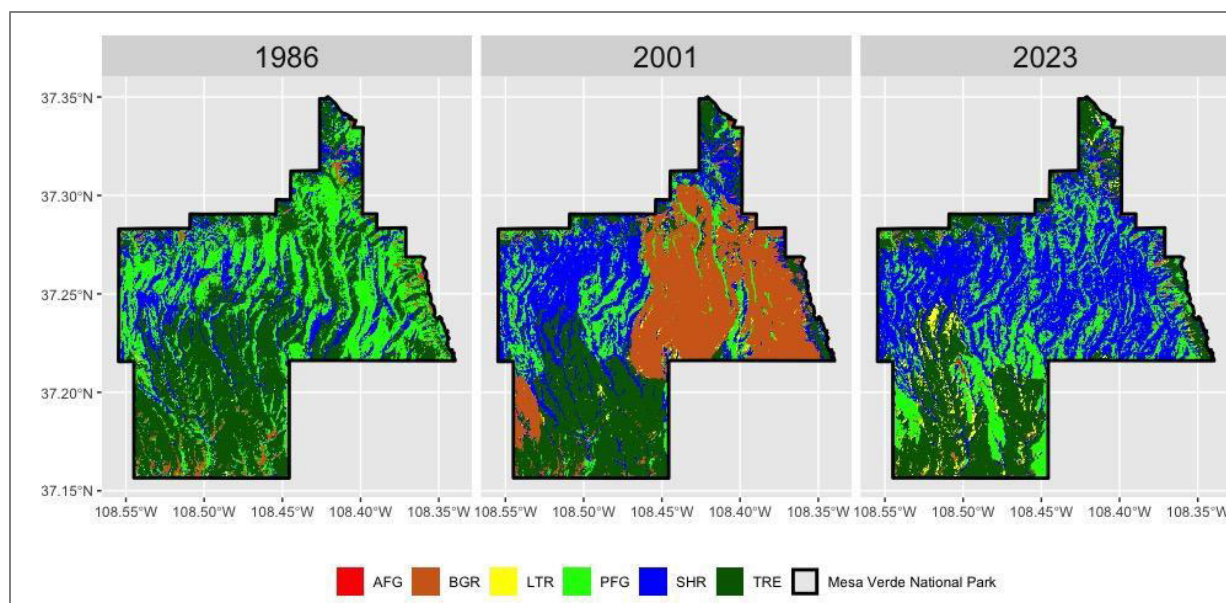


Figure 35. Categorical change in dominant vegetation type (AFG: annual forbs and grasses; BGR: bare ground; LTR: litter; PFG: perennial forbs and grasses; SHR: shrub; TRE: tree. over time: 1986¹⁰, 2001 (post major fires) and 2023. Data generated from RAP fractional cover, here illustrating the plurality cover class at 30 m resolution across MEVE. Created using data from Rangeland Analysis Platform (Allred et al. 2021).

NPS / CONSERVATION SCIENCE PARTNERS

¹⁰ Data from 1986 was used because this was as far back as the data went; the mid-1980s should not be inferred as a “baseline” of ecosystem integrity at MEVE (i.e., in the 1950s the park seeded smooth brome in the northern part of the Park after fires. Also the 1980s were a wet period for MEVE).

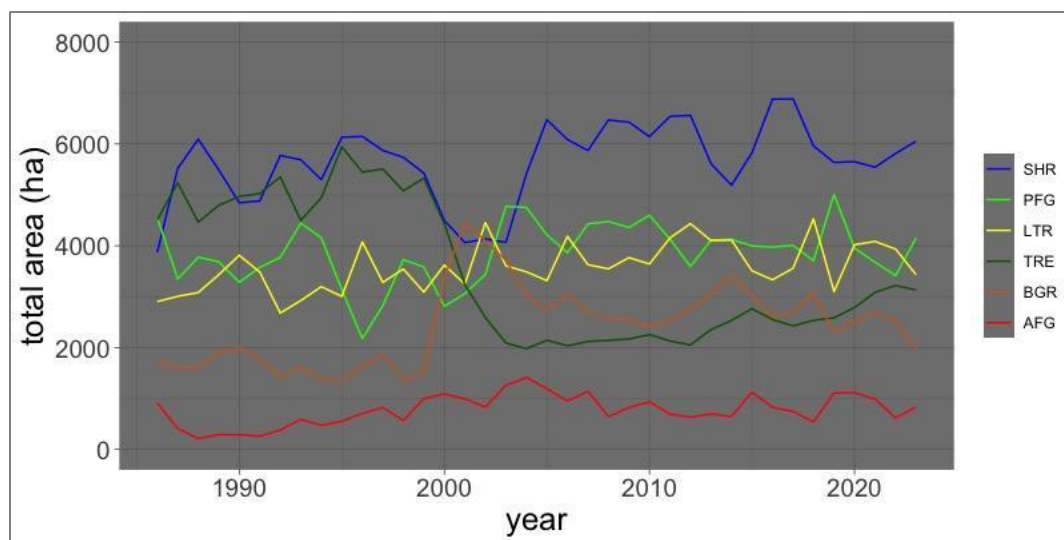


Figure 36. Computed aggregate area by cover class per year (area of each pixel multiplied by fractional cover values) for shrubs (SHR), perennial forbs and grasses (PFG), litter (LTR), trees (TRE), bare ground (BGR), and annual forbs and grasses (AFG) in Mesa Verde National Park. Data generated from RAP fractional cover at 30 m resolution across MEVE. Created using data from Rangeland Analysis Platform (Allred et al. 2021).

NPS / CONSERVATION SCIENCE PARTNERS

Condition of and Change in Habitat Structure to Support PJ and Shrub Associated Passerines

The state of the passerine community can confer insight to the condition of the woodland and shrubland community at MEVE. As described earlier in the Passerine Condition Assessment Summary, one potential outcome of increased fire and eventual transition to shrub habitat (in place of PJ) is that it can favor certain MEVE bird species. For example, our model showed that nine (64%) of the MEVE bird species analyzed increased in density with “deciduousness” (or non-PJ, shrub habitat) (Figure 12 in avian section). This suggests that some species may be able to adapt to post-fire landscapes as succession unfolds—though it should be noted that how the habitat recovers after fire may be an important factor in these species’ persistence. Conversely, five (35%) of the MEVE birds analyzed exhibited statistically significant decreases in density immediately following the fire. We attribute this decline to the loss of key resources and habitat features essential for some species, as structural and ecological heterogeneity diminishes in the aftermath of the fire.

Long-Term Post-Fire Monitoring. The Pony Fire in 2000 and the Long Mesa Fire in 2002 collectively burned 7048 acres of Park land (see Figure 34), with the majority of burned area being PJ woodlands. The burned landscapes from both fires were aerially seeded with grass in the hopes of stabilizing slopes and preventing the establishment of non-native species. In 2011, botanists developed the long-term post-fire monitoring program (LTPFMP) to measure vegetation changes and understand how burned woodlands recover after major fire events. Researchers established 53 plots that had burned in one of the two fires and collected data in 2011, 2012, 2013, 2014, and 2019.

Researchers collected data on native forb cover, native shrubs, seeded species, non-native species, PJ seedlings and recruitment, and biological soil crust development. Reports from the earlier

assessments (Garmoe and Perry 2014; Wender and Doherty 2011, 2013) indicate no PJ recruitment, low coverage of native shrubs, high coverage of native forbs, and relatively high coverage of native perennial grasses, as well as non-native species such as cheatgrass (*Bromus tectorum*), musk thistle (*Carduus nutans*), and western wheatgrass (*Pascopyrum smithii*) [note: while western wheatgrass is native to the Colorado Plateau, it is an “offsite species that has become invasive in PJ at MEVE” (Wender and Doherty 2011, 2013)]. The maximum level of development of biological soil crust was low in both fire sites (Garmoe and Perry 2014). In 2019, sites from both fires were still in the early stages of succession with minimal PJ recruitment. Of the 27 PJ seedlings found in 2013, only 4 remained by 2021 (Knight et al. 2021). Other findings from 2019 included a high establishment of native forb species, increased coverage of one native shrub species (Rubber rabbitbrush [*Ericameria nauseosa*]), and increased coverage of one non-native grass species (cheatgrass). No data were collected on biological soil crust development for the 2019 report.

During the initial years of reorganization, burned areas may be dominated by grasses, forbs, and sprouting shrubs, with later successional stages dominated by trees, shrubs, and perennial grasses (Erdman 1970; Floyd et al. 2000a; Koniak 1985). We observed similar patterns in MEVE via the LTPFMP. Burned landscapes in the earlier years were predominantly native and non-native grasses, with native shrubs becoming more prevalent almost 20 years post-fire. Rondeau (2023) observed an increase in native shrubs in most Long Mesa Fire plots in 2022, with coverage as high as 35%. The regeneration of PJ can be an extremely long process, and after stand-replacing fires, it could take decades for new trees to re-establish (Floyd et al. 2004, 2000a; Thomas et al. 2009). Though the lack of tree regeneration at MEVE is not abnormal, it has been speculated that the heavy presence of non-native species and grasses, such as western wheatgrass, and changing climatic conditions may make piñon recruitment more difficult (Floyd et al. 2003; Spector et al. 2019). Juniper recruitment may not be so affected.

Phillips et al. (2023) studied the vegetation structure, microclimatic conditions, and PJ regeneration dynamics of sites burned in the Long Mesa and Pony fires and compared system changes to those of intact and thinned PJ woodlands within MEVE and on nearby Ute Mountain Ute Tribal Lands. They concluded it was unlikely that MEVE’s PJ would recover passively following both fires for several reasons. First, increased grass cover (seeded natives and non-natives) in burned areas increases a landscape’s disposition for future fire due to the high flammability of grasses and decreased interspace size between plants (Balch et al. 2013; Bradley et al. 2018; Floyd et al. 2021; Fusco et al. 2019). Second, climatic changes causing lower minimum daily vapor pressure and higher photosynthetically active radiation may result in lower soil moisture in burned areas than in intact PJ woodlands, thereby exposing potential seedlings to drier conditions that inhibit seed germination and seedling establishment (Phillips et al. 2023). Third, the biocrust communities of burned landscapes differ from intact communities (Phillips et al. 2023), which may result in decreased soil stability (Belnap and Gillette 1997) and fertility (Ferrenberg et al. 2018) in burned landscapes.

Most of the climatic and soil properties considered by Phillips et al. (2023) were not in the latest LTPFMP for 2019 (Spector et al. 2019). However, the post-fire invasion of grasses coupled with climate change are creating conditions seemingly inhospitable for PJ regeneration, possibly forcing a

state change from PJ woodlands to one dominated by perennial grasses with sparse shrub cover (Phillips et al. 2023). Indeed, climatic conditions are projected to become marginal or unsuitable for certain PJ species across much of their present range (Rondeau et al. 2011).

Non-native Plant Summarizations. Stand-replacing fires, including the 1989 Long Mesa Fire and the 2003 Balcony House Complex Fire, have led to an increase in non-native vegetation (e.g., annual forbs and grasses) within MEVE. Annual forb and grass (AFG) data from the Rangeland Analysis Platform (Allred et al. 2021) from 1987 to 2023 in MEVE were acquired. Generalized Least Squares (GLS) models were applied to prior/during stand-replacing fire (1987–2004) data and post stand-replacing fire (2005–2023) data separately. The GLS model indicated a significant positive effect of year on annual forb and grass cover for the period prior/during stand-replacing fire (1987–2004) ($\beta_1 = 0.2909$, $p\text{-value} = 0.0$), suggesting a positive trend in percent cover of AFG during this period (Figure 37). Assumptions were tested and residuals were normally distributed via a Shapiro test ($p\text{-value} > 0.05$) and no significant autocorrelation was detected (Ljung-Box test $p\text{-value} > 0.05$). There was no statistically significant trend found for the post-fire AFG percent cover data ($\beta_1 = 0.0289$, $p\text{-value} = 0.4885$) (Figure 37). After these major fires, the percent cover of annual forbs and grasses remained stable within MEVE but was higher than pre-fire levels (Figure 37). Non-natives likely increased post-fire because they have a faster rate of recovery than native species, conferring a competitive advantage with the availability of bare soil post-fire (Floyd et al. 2006). Further details on the GLS analysis can be found in Appendix B.7.

In general, there was a higher density of non-native species, including musk thistle and cheatgrass, within burned MEVE landscapes compared to unburned landscapes (Floyd et al. 2006, 2021). Floyd et al. (2021) found significantly lower densities of these non-native species in burned areas of mountain shrub communities compared to burned PJ habitats. This is because MEVE shrub communities contain sprouting shrub species (as opposed to seedlings, characteristic of PJ woodlands), and sprouting species have been shown to be more effective at recolonizing and competing in disturbed landscapes than weeds (Floyd et al. 2021). Floyd et al. (2006) also developed a spatial model to predict areas in MEVE at risk of future invasion by non-native plants after fires. They determined that approximately 35% (7,512 ha) of MEVE was classified as “high-risk”, and 19% (4,055 ha) at “moderate risk”, from non-native species post-fire. The at-risk areas occur primarily in the southern portions of the Park, where old-growth PJ woodlands are dominant (Floyd et al. 2006).

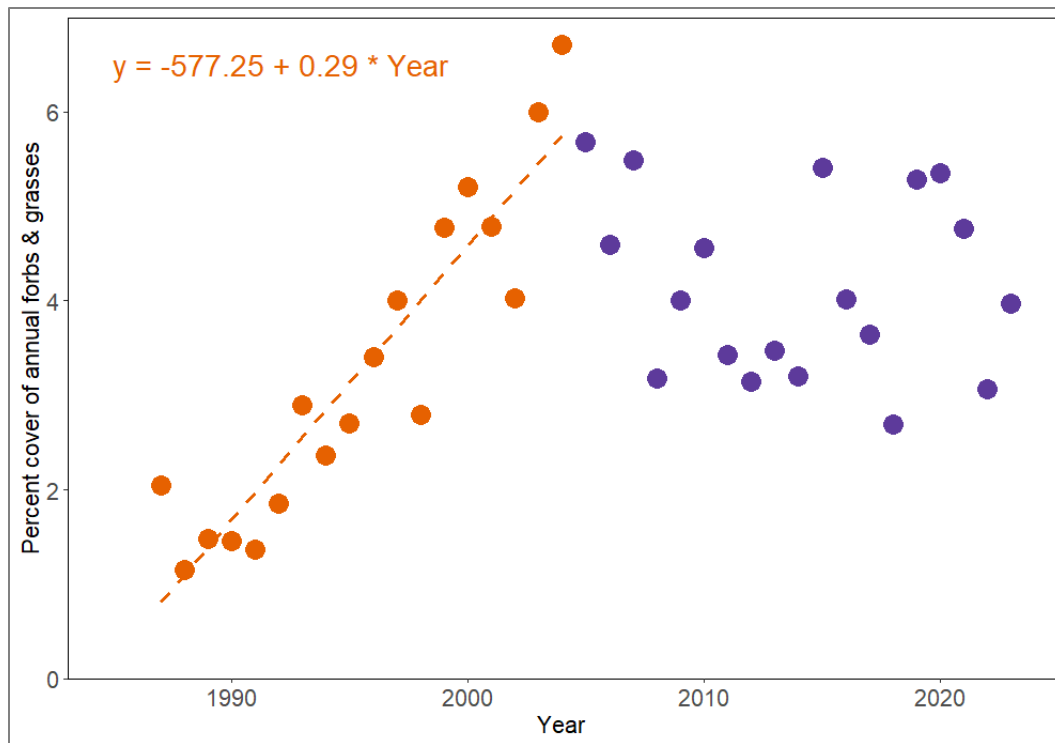


Figure 37. Percent cover of annual forbs and grasses within MEVE from 1987 to 2023. The time series is split where 1987–2004 (orange circles) represents before and during stand-replacing fires, and 2005–2024 (purple circles) represents after stand-replacing fires. The GLS model's fitted values from the prior/during stand-replacing fire data are represented as a trendline. The associated equation of the trendline is also reported. The post stand-replacing fire GLS model was not reported. Created using data from Rangeland Analysis Platform (Allred et al. 2021).

NPS / CONSERVATION SCIENCE PARTNERS

MEVE Park Staff provided all data on direct observations and mapping of non-native species in the Park, which we utilized in our analyses of non-native plant treatments. Documentation of non-native plant treatments, including spatial polygons of treatment location and type (Figure 38), spans from 2012 to 2023. Non-native plant treatments include chemical (herbicides sprayed via backpack/spot sprayer or UTV sprayer), manual (pulled plants and roots), and mechanical mechanisms (removed by cutting via weed whackers, mowers, trimmers, saws, etc.). Chemical treatments are often the most effective way of treating non-natives, as they require less time and personnel than manual and mechanical treatments (Stowell 2022b). Non-native treatments typically occur near roads and infrastructure such as the MEVE Visitor Center and the Morefield Campground (Figure 38). Recent treatments typically focused on these areas because high disturbance perpetuates non-native growth, and a smaller budget required a prioritization of areas with high visitation versus remote locations (Stowell 2022b). Park staff treated high disturbance areas such as the Visitor Center and campground as recently as 2023; however, roads were last treated on a large scale in 2019 and 2021.

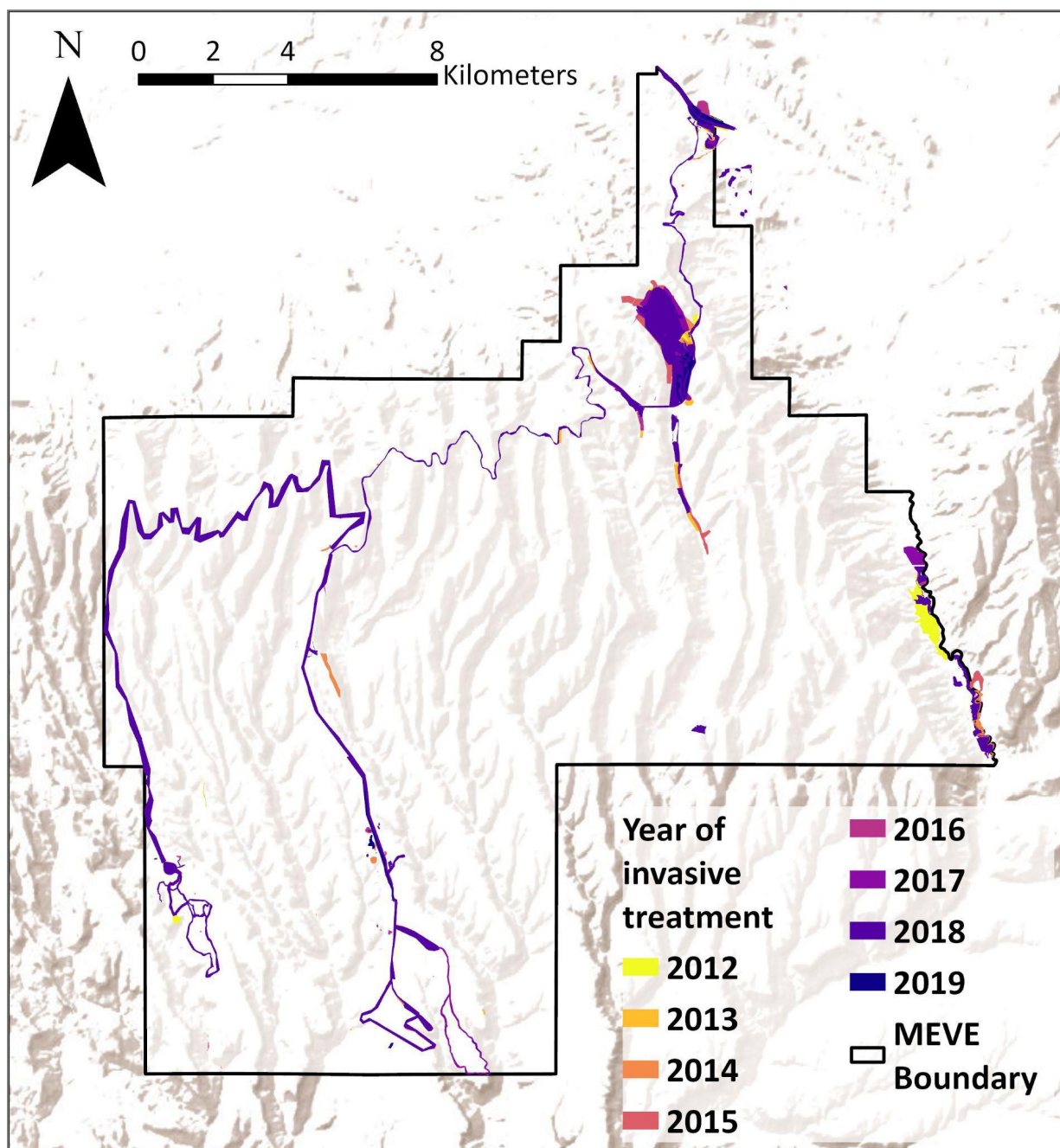


Figure 38. Distribution of non-native plant treatments from 2012 to 2023. Created using data from MEVE staff.

NPS / CONSERVATION SCIENCE PARTNERS

In 2021, a shift in Park software resulted in a new non-native treatment protocol developed by Park staff (personal communication, Nikita Prasser, MEVE, December 2023). This change in methodology has led to variations in data documentation between 2012–2021 and 2022–2023. Additionally, there were significant fluctuations in values between years, particularly in the volume of chemicals sprayed, and often a lack of information about the specific units of chemical applied.

Consequently, we were only able to report the annual acres of treatments (Figure 39). Due to a lack of differentiation between treatment types, mechanical and manual had to be combined. Among the documented years of non-native treatments, 2018 by far had the highest acreage of chemical and mechanical/manual treatments. The limited treatments in 2022 were due to budget limitations, with an increase in treatments in 2023.

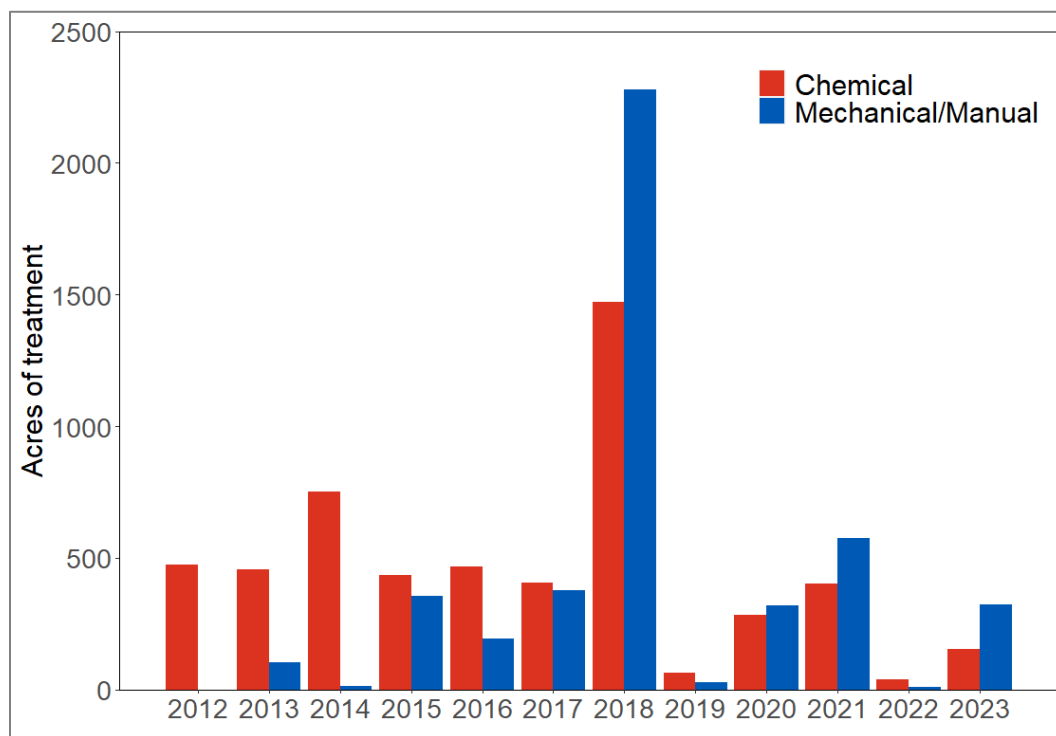


Figure 39. Total acreage of chemical and mechanical non-native plant treatments within MEVE from 2012 to 2023. Created using data from MEVE staff.

NPS / CONSERVATION SCIENCE PARTNERS

Piñon-Juniper (PJ) Woodlands Condition Assessment Summary

Summary of SCPN Vegetative Indicators from PJ Plot Data

The Southern Colorado Plateau Network (SCPN) monitored vegetation and soils in piñon-juniper woodlands in MEVE from 2008 to 2022 (Swan 2024). SCPN staff established 30 long-term monitoring plots in two PJ Ecological Sites (hereafter, ecosites) in MEVE: Loamy Mesa Top Piñon Juniper Woodland (ESD R036XY142CO), and Shallow Loamy Mesa Top Piñon Juniper Woodland (ESD R036XY141CO) (DeCoster et al. 2012) (Figure 40). Both ecosites feature old-growth PJ woodlands dominated by Utah juniper and two-needle piñon (Swan 2024), and consist of relatively large, intact sites that are highly valued by Park staff and regional experts (Thomas et al. 2006). Swan (2024) reported that these MEVE ecosites should be classified as persistent, old growth PJ woodlands, with limited understory vegetation, very old trees (> 400 years old) and historically long fire-return intervals. To establish the monitoring plots within these two ecosites, SCPN followed standard SCPN protocol used at other sites. 30 plots were monitored within each of the two ESDs.

Monitoring began at the Loamy Mesa Top plots (or “Loamy plots”) in 2007, and the Shallow Loamy Mesa Top plots (or “Shallow Loamy plots”) in 2012. Monitoring plots were 50 m × 52 m and consisted of three parallel 50 m transects spaced 25 m apart (DeCoster et al. 2012). A subset of the plots was visited each year, except for 2020 (see Appendix A.1 in Swan (2024) for specific plots sampled each year). Plots were sampled as close to peak phenology as possible and were sampled at approximately the same time each year (Swan 2024).

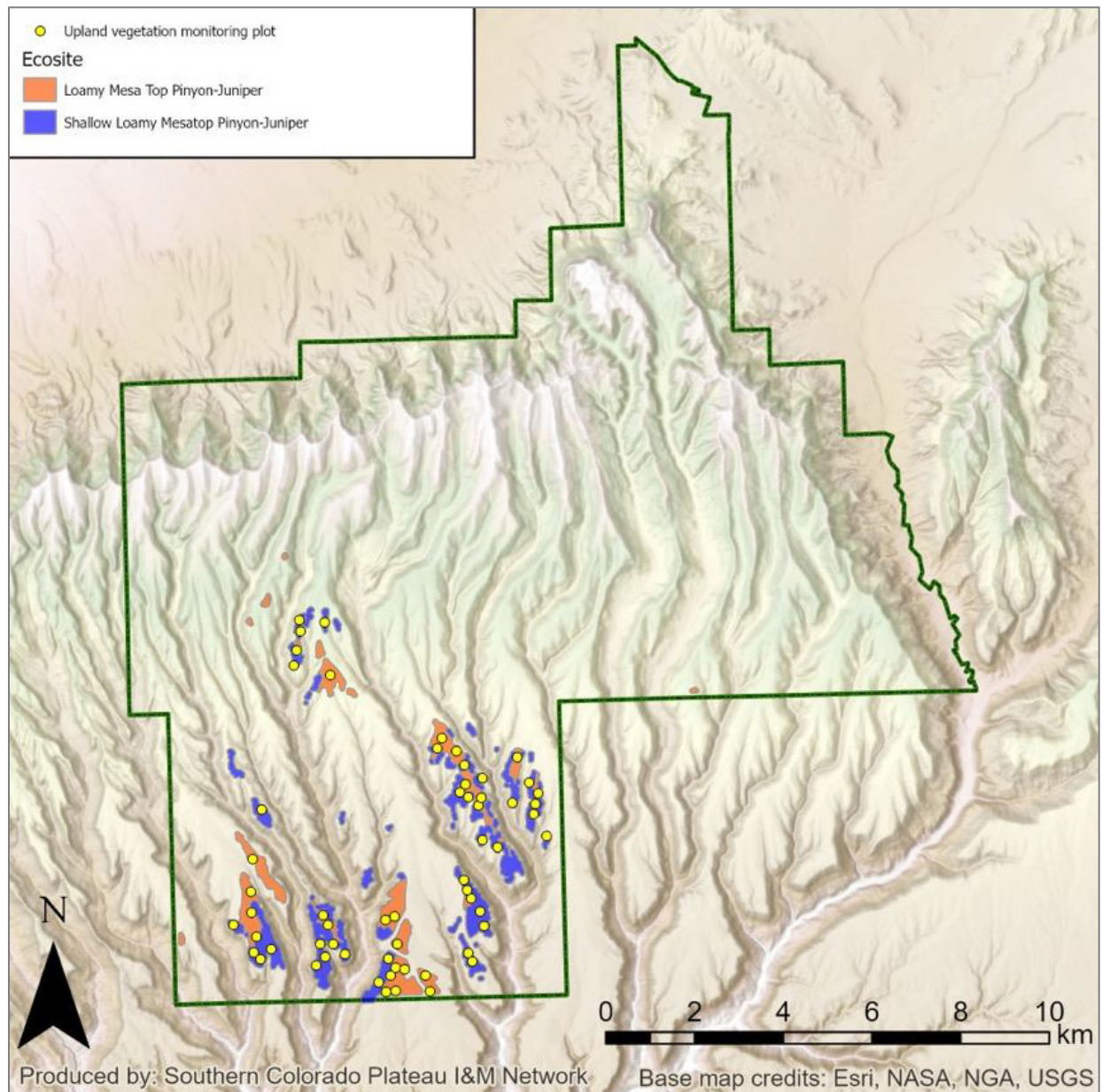


Figure 40. Overview map of the SCPN sampling frames and plot locations within the PJ Ecological sites in MEVE.

NPS / SOUTHERN COLORADO PLATEAU NETWORK

SCPN data on PJ seedling recruitment, overstory cover of PJ, and cover of shrubs, perennial grass, annual grass and forbs were summarized at the plot level, while plant species richness was quantified at three scales (10 m² quadrat, 50 × 52 m plot, entire ecosite). Data are presented as averages across all survey years (2008–2012), thus reducing the influence of inter-annual variability.

Swan (2024) calculated species richness of all (non-tree) plants at the two ecosites at the quadrat and plot scales (Table 10). Swan also calculated average percent cover (of total ground cover) of perennial grass at both ecosites and found that the Loamy Plots averaged 3.6% total cover (which was 42% of all foliar cover) of perennial grass, and the Shallow Loamy Plots averaged 2.8% total cover (or 27% of all foliar cover) of perennial grass (Table 10). Muttongrass (*Poa fendleriana*) was the dominant grass in both ecosites, occurring in over 90% of plots (Swan 2024). Squirreltail (*Elymus elymoides*) was also common. Forbs occurring in the SCPN plots represented a mix of annual, biennial, and perennial species with varied growth forms. Forbs comprised 1% of total cover at the Loamy Plots (or 11.8% of all foliar cover) and 0.6% of total cover (5.8% of all foliar cover) at the Shallow Loamy Plots. Toadflax penstemon (*Penstemon linarioides*) was the most common forb detected; it occurred in 79% of Loamy Plots and 51% of Shallow Loamy plots. At the Loamy plots, shrubs comprised 2.8% of total ground cover (nearly 33% of all foliar cover), and at the Shallow Loamy plots, shrubs comprised 5.7% of total ground cover (over 55% of all foliar cover). Annual grass (which was only cheatgrass) at both ecotypes averaged much less than 1% of both total ground cover and percent foliar cover (Swan 2024) (Table 11).

Table 10. Average quadrat (10 m²) and plot-scale (1500 m²) richness of all (non-tree) plants and range (lowest and highest values) for Loamy and Sandy Loamy ecological sites in MEVE during 2008–2022. Table derived from (Swan 2024).

Richness Type	MEVE Loamy Plots	MEVE Shallow Loamy Plots
Median quadrat richness (range)	6 (1,16)	6 (0,16)
Median plot richness (range)	18 (9,34)	21 (9,36)

Table 11. Average plot-level percent cover by lifeform in two ecosites in MEVE during 2008–2022. Values in parentheses represent lifeform percent cover as a percentage of total live foliar cover. Table derived from (Swan 2024).

Life Form	MEVE Loamy Plots	MEVE Shallow Loamy Plots
Total live foliar cover	8.5	10.3
Perennial grass	3.6 (42.3%)	2.8 (27.2%)
Shrub	2.8 (32.9%)	5.7 (55.3%)
Forb (not incl. cactus/succulents)	1.0 (11.8%)	0.6 (5.8%)
Annual grass	0.01 (0.12%)	0.01 (0.1%)

Both piñon and juniper seedlings were recorded by seedling size class at the two ecosites from 2008 to 2022 (Swan 2024) (Figure 41). However, to assess the condition of this indicator as guided by the

literature, we focused solely on piñon seedlings (all size classes) per m². The average number of piñon seedlings found at the Loamy plots was 0.87 seedlings/m², and the average number of piñon seedlings found at the Shallow Loamy plots was 0.79 seedlings/m² (Swan 2024).

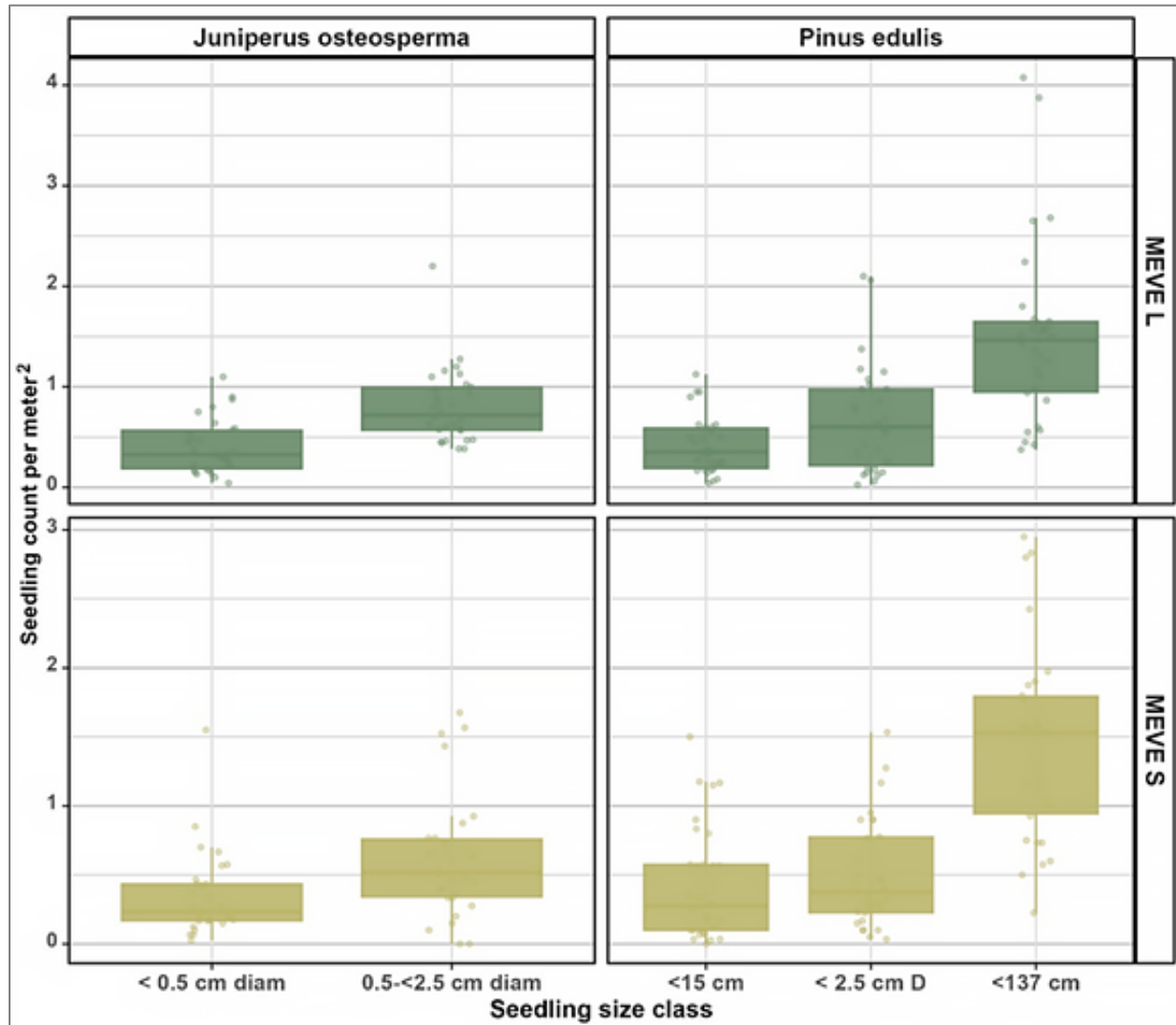


Figure 41. Mean tally of seedlings by species and size class at the Loamy (L) and Shallow Loamy (S) ecosites in MEVE during 2008–2022. Colored boxes indicate the 25th and 75th percentiles, and the vertical lines indicate the range of percent cover (excluding outliers) for each surface feature. The horizontal line indicates average seedling count per meter squared. Colored dots show plot-level means. NPS / SCPN

Swan (2024) also reported total canopy cover of all trees (PJ) at the two ecosites; average canopy cover of trees was 34% at the Loamy Plots and 39% at the Shallow Loamy Plots (Figure 42).

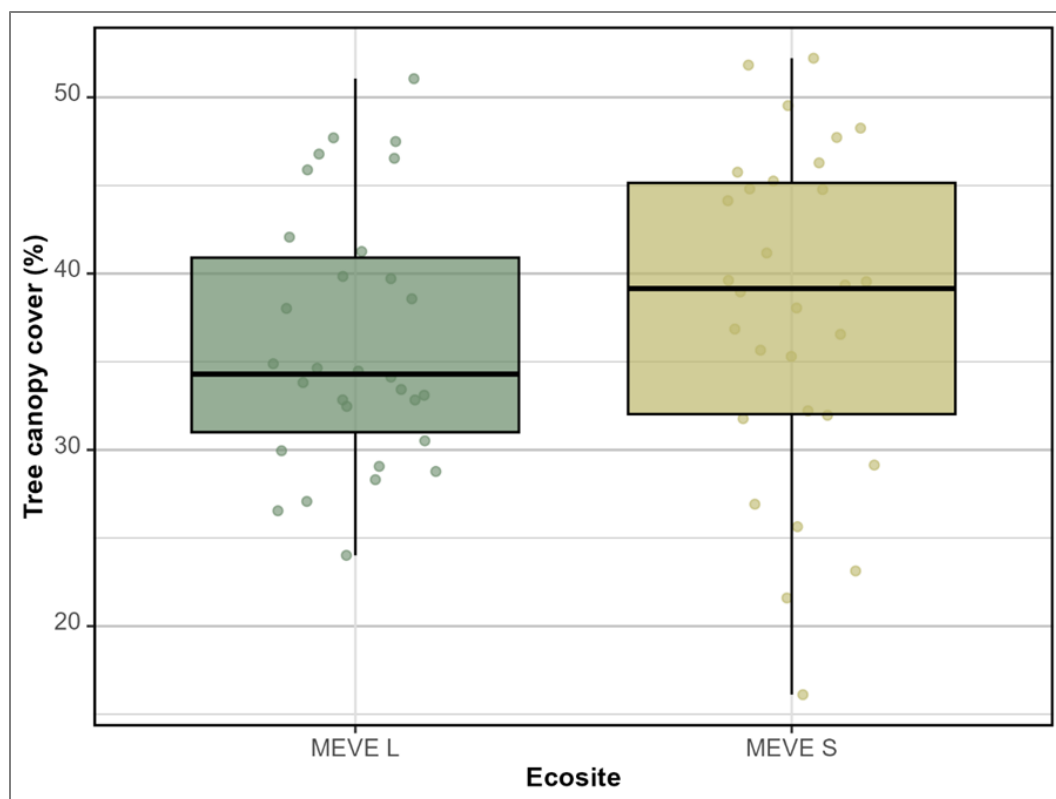


Figure 42. Mean percent canopy cover at the MEVE Loamy PJ Woodland ecosite (left) and the Shallow Loamy ecosite (right) during 2008–2022. Colored boxes indicate the 25th and 75th percentiles, and the vertical lines indicate the range of percent canopy cover. The horizontal line indicates the median percent canopy cover. Colored dots indicate plot-level means.

NPS / SCPN

Assessing Trends of PJ Condition Indicators

NPS Inventory and Monitoring Networks are tasked with reporting on status and trend of selected natural resources (Thomas et al. 2006). While significant change in vegetation communities commonly occurs over decades or centuries, some metrics (e.g., richness, plant cover) can exhibit high variability from year to year in response to variable drivers such as temperature and rainfall (Dios Miranda et al. 2009). To determine trends in SCPN’s MEVE plot variables over time, Swan (2024) used a Bayesian hierarchical modeling approach to investigate changes over time in plant cover, quadrat-scale richness, and PJ seedling and sapling numbers as a function of climatic water deficit (the difference between potential evapotranspiration and actual transpiration) and cumulative growing degree-days (January–June).

Swan (2024) modeled 2008–2022 perennial grass, shrub, and forb responses to previous year water deficit, winter deficit, spring deficit, and aggregated growing degree days (AGDD), Aridity index (AI), and total plot basal area of overstory trees and terrain across the two MEVE ecosites. Results of the perennial grass model, despite considerable spatial variability in grass cover between quadrats and considerable uncertainty even with 15 years of data, (Swan 2024), indicated that perennial grass cover decreased over the monitoring period at the Loamy plots and increased at the shallow loamy

plots (Swan 2024). For shrubs, the modeled trend from 2008 to 2022 was not significant at the Loamy plots, while the Shallow Loamy plots showed a positive trend in overall shrub cover (Swan 2024). It is perhaps intuitive that MEVE trends in shrub cover within PJ stands were either neutral or positive; shrubs are longer-lived than herbaceous species which results in slower changes in cover in response to relatively short-term fluctuations in conditions compared to herbaceous species (Flake and Weisberg 2021). Since forbs were a minor component of overall vegetation in both of the MEVE ecosites, Swan (2024) did not run a forb-specific Bayesian model, instead they modeled toadflax penstemon trends, because this forb occurred in 78% of the Loamy plots and 51% of the Shallow Loamy plots. Swan (2024) found that presence of toadflax penstemon increased at the Shallow Loamy ecosite and did not change significantly at the Loamy ecosite. Winter water deficit had a positive effect on presence of this dominant forb, indicating that toadflax was encountered more frequently in years with higher winter deficits.

To determine trends in species richness, the Bayesian model included the covariates' lag year, winter and spring water deficit, an Aridity index, and annual growing degree days (Swan 2024). Swan found that species richness increased over the monitoring period at both MEVE ecosites.

For PJ indicators, Swan combined the seedling size classes and modeled the density of PJ seedlings across the two MEVE ecosites. The models included previous-year water deficit, winter deficit, and spring deficit (Swan 2024). Both juniper and piñon seedling density increased significantly from the 2008–2022 period at the Loamy plots, while there were no significant trends at the Shallow Loamy plots. Swan (2024) also modeled changes in PJ canopy cover over time at MEVE and found no significant trends. Swan (2024) also modeled mortality of overstory trees, and found that it was not widespread during the monitoring period. In fact, about 90% of trees did not change status between their initial assessment during the first visit to a plot and the most recent visit (Swan 2024). The maximum number of individual trees that died within a MEVE single plot was 21 at Loamy Plot #29, which experienced wildfire in August 2018.

SCPN PJ Plots in context of other SCPN monitoring sites and climate change

Swan (2024) additionally analyzed the MEVE plot data with similar plot data in PJ Ecosites at Bandelier National Monument (BAND) and Grand Canyon National Park (GRCA). In BAND and GRCA, perennial grass cover did not change significantly from 2008 to 2022 (Swan 2024). For all vegetation cover variables modeled across the three park units (MEVE, BAND and GRCA), Swan (2024) reported that spring climatic water deficit had the strongest overall negative effect on vegetation cover. Overall grass and shrub cover, as well as cover of many of the species modeled, decreased in response to higher spring water deficit (Swan 2024). This is concerning because semi-arid woodlands are expected to be affected by severe drought driven by predicted increasing temperature ($>2^{\circ}\text{C}$ in the near-term future) and decreasing soil moisture availability, especially in the spring season (Andrews et al. 2019; Bradford et al. 2020), as cited in (Swan 2024). Swan (2024) noted that during the monitoring period (2008–2022) the southwest United States was in a period of historic drought. 2000–2021 was the driest two-decade since at least 800 CE, and possibly longer (Williams et al. 2022). Swan (2024) reported that the driest conditions relative to normal for the

MEVE plots were experienced in 2018 when the MEVE ecosites received on average < 50% of normal precipitation for the year.

While some plots are accessible to and potentially impacted by feral horses, Swan (2024) concluded the SCPN PJ monitoring report by noting that, “The sites included in this report, as part of the National Park Service, are almost uniquely protected from most localized anthropogenic disturbances, [including] domestic grazing ... and many other non-climate related drivers of change that contribute to change or mortality of vegetation. Therefore, it is highly likely that change in these sites is being driven by changing climate. Further refining models and predictor variables, as well as additional years of monitoring data will help clarify these relationships.”

Condition Assessment for PJ woodlands in MEVE

To complete the condition assessment for PJ habitat condition in the Park, the current condition status of indicators of species richness, piñon seedling recruitment, PJ canopy cover and cover of shrubs, perennial grass, forbs and annual grass were assessed using the PJ condition assessment rating thresholds developed in consultation with Megan Swan of the SCPN (Appendix A.7). Thresholds for the rating condition categories were in some cases based on the Ecological Site Descriptions for Loamy Mesa Top piñon Juniper Woodland (ESD R036XY142CO), and Shallow Loamy Mesa Top piñon Juniper Woodland (ESD R036XY141CO), and in some cases, the literature. Table 12 below illustrates the outcome of this exercise. We believe we have achieved a high confidence level in these condition rating results, given that the values that went into the condition rating table were derived directly from SCPN’s 2024 PJ Monitoring report for MEVE (Swan 2024).

Table 12. Piñon-juniper habitat condition ratings summary. Condition ratings below are based on the Condition Rating Table in Appendix A.

Indicator	Condition	Trend	Condition Evaluation
Shrub Cover - Loamy PJ Ecosite	Good	Stable	The condition of shrub cover in Loamy PJ Ecosites is good because cover in these plots was over 20% of all foliar cover.
Shrub Cover - Shallow Loamy PJ Ecosite	Good	Upward	The condition of shrub cover on Shallow Loamy PJ Ecosites is good because cover in these plots was over 20% of all foliar cover.
Perennial Grass Cover - Loamy PJ Ecosite	Good	Downward	The condition of perennial grass cover in Loamy PJ Ecosites is good because cover in these plots was over 15% of all foliar cover.
Perennial Grass Cover - Shallow Loamy PJ Ecosite	Good	Upward	The condition of perennial grass cover in Shallow Loamy PJ Ecosites is good because cover in these plots was over 15% of all foliar cover.
Forb Cover - Loamy PJ Ecosite	Good/Fair	Stable	The condition of forb cover (for Toadflax penstemon, most common forb) in Loamy PJ Ecosites is good/fair because cover in these plots was 11.8% of all foliar cover.
Forb Cover - Shallow Loamy PJ Ecosite	Fair	Upward	The condition of forb cover (for Toadflax penstemon, most common forb) in Shallow Loamy PJ Ecosites is fair because cover in these plots was only 5.8% of all foliar cover.

Table 12 (continued). Piñon-juniper habitat condition ratings summary. Condition ratings below are based on the Condition Rating Table in Appendix A.

Indicator	Condition	Trend	Condition Evaluation
Annual Grass Cover - Loamy PJ Ecosite	Good	N/A	The condition of annual grass cover (cheatgrass) in Loamy PJ Ecosites is good because cover in these plots was <1% of all foliar cover.
Annual Grass Cover - Shallow Loamy PJ Ecosite	Good	N/A	The condition of annual grass cover (cheatgrass) in Shallow Loamy PJ Ecosites is good because cover in these plots was <1% of all foliar cover.
Plot Species Richness - Loamy PJ Ecosite	Good/Fair	Upward	The condition of species richness in Loamy PJ Ecosites is good/fair because 18 species of plants were found in the 1500 m ² plots.
Plot Species Richness - Shallow Loamy PJ Ecosite	Good	Upward	The condition of species richness in Shallow Loamy PJ Ecosites is good because 21 species of plants were found in the 1500 m ² plots.
Pinyon Seedling Recruitment - Loamy PJ Ecosite	Fair	Upward	The condition of pinyon seedling recruitment in Loamy PJ Ecosites is fair because seedling density was 0.87 seedlings/m ² in these plots.
Pinyon Seedling Recruitment - Shallow Loamy PJ Ecosite	Fair/Poor	Stable	The condition of pinyon seedling recruitment in Shallow Loamy PJ Ecosites is fair/poor because seedling density was 0.79 seedlings/m ² in these plots.
PJ Canopy Cover - Loamy PJ Ecosite	Good/Fair	Stable	The condition of PJ canopy cover in Loamy PJ Ecosites is good/fair because PJ canopy cover was 34% in these plots.
PJ Canopy Cover - Shallow Loamy PJ Ecosite	Good/Fair	Stable	The condition of PJ canopy cover in Shallow Loamy PJ Ecosites is good/fair because PJ canopy cover was 39% in these plots.

3.6 Seeps and springs

Desert springs are renowned as hot spots of biological and cultural significance (Fensham et al. 2023; Stevens et al. 2020). MEVE springs are typically small (average discharge less than 0.2 L/sec) but are ecologically important micro-communities that are fed by the Park's several aquifers. Water quality at MEVE springs varies, but the average water temperature is about 13°C, and all are cool-water springs. MEVE spring water is often naturally high in calcium carbonate, sulfate, sodium, and manganese concentrations, and some sites have elevated concentrations of aluminum and boron.⁴

While rheocrene and hillslope spring plant assemblages in MEVE are dominated by a mixture of relatively common riparian species, hanging gardens create unique habitats that support regionally rare plant species, including sedges and ferns. Hanging gardens support important habitats such as shaded, wet backwalls that feature the most distinctive vegetation assemblage of all southwestern spring types (Sinclair 2018). These habitats are sensitive to climate change-induced warming, decreased precipitation, and degree of groundwater recharge (Weissinger et al. 2016; Welsh and Toft 1981). Among several springs for which data are available in MEVE, wetland/riparian vegetation is dominated by *Carex* and *Schoenoplectus* sedges, *Deschampsia* and several other grasses, and herb

species. Wetland-riparian species include native common reed (*Phragmites australis americana*), shrub coyote willow (*Salix exigua*) and Goodding's willow (*S. gooddingii*), and virgin's bower (*Clematis* spp.). Of the rare and endemic species currently tracked by MEVE (see Table 3 above), Giant hellebore (*Epipactis gigantea*) is the only one likely to be found in spring systems and might experience declines in the case of spring drying. Another riparian/wetland plant species that in the future might be added to the MEVE tracking list is the maiden hair fern (*Adiantum capillus-veneris* L., G5 S2). This species currently only exists at one spring site in MEVE and has declined in health with the spring being more routinely dry in recent years (personal communication, Andrew Spear, Physical Science Program Manager, MEVE, November 2024). A large number of non-native plant species also occur in and around springs in MEVE, including non-native grasses, thistles, and tumbleweed.

Springs support important wildlife habitat. For example, many of the Park's common bird and mammal species have been detected at its springs, including top predators such as peregrine falcons, golden eagles, bobcats (*Lynx rufus*), and American black bears. Although only a few of MEVE's springs have been inventoried thoroughly, they are clearly high-value habitats that support disproportionately high and unique concentrations of the Park's flora and fauna. While livestock grazing is not allowed in MEVE, the few trespass cattle that enter MEVE often concentrate at water sources, significantly damaging them by trampling and grazing (Perry et al. 2015).



Southern maidenhair fern at Fewkes Canyon Spring #4.

NPS

Seeps and Springs Gap Analysis Summary

MEVE Park staff provided spatial data on 265 seeps and springs (hereafter referred to as springs) within and nearby MEVE (Figure 43). The spatial data did not include all springs documented in MEVE's master database. To estimate the degree of human disturbance to springs, we calculated the number of springs that were within 10 m of a road or hiking trail, with 10 m representing the minimum distance required to safeguard spring properties (particularly water quality) from

anthropogenic disturbance¹¹ (Clinton 2011; Parkyn 2004). Only eleven of the 265 springs (~4%) were within 10 m of a main road, backcountry road or hiking trail¹².

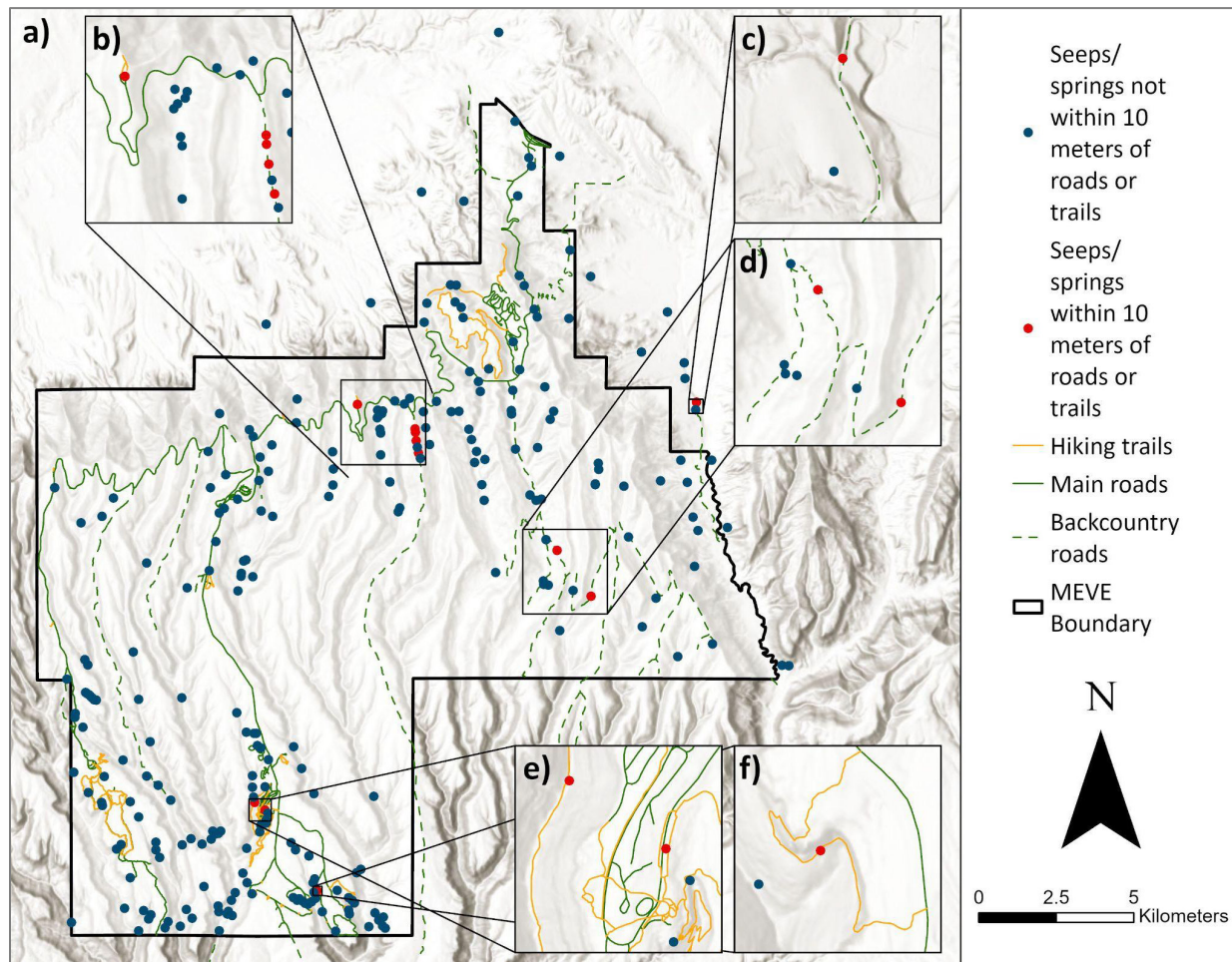


Figure 43. a) Distribution of 265 springs within and nearby MEVE and their relationship to human disturbance in the form of trails and roads. Panels b, c, d, e, and f are zoomed in locations of the 11 springs that were within 10 meters of roads and hiking trails. Created using springs & road data from MEVE staff and hiking trails data from National Park Service (2017).

NPS / CONSERVATION SCIENCE PARTNERS

¹¹ Despite a backcountry closure in 1996 restricted widespread trail use, there are a few springs that are heavily impacted from trail use and tours, especially springs close to popular archeological sites (personal communication, Andrew Spear, Biologist, MEVE, November 2024).

¹² Park Point Cistern #269 (last observed [l.o.] 1994) (Figure 43b) and Spruce Tree Canyon Head #141 (l.o. 2011) (Figure 43e) were within 10 meters of a main road. All four Moccasin Mesa Road Puddles #157-5 (l.o. 1996) (Figure 43b), Buffaloberry Draw #265 (l.o. 1994) (Figure 43c), Moorefield Ridge Spring #165 (l.o. 2014) (Figure 43d), and Waters canyon - white mesa road puddle #156 (l.o. 2001) (Figure 43d) were within 10 meters of a backcountry road. Spruce Canyon Trail Spring #1004 (l.o. 2023) (Figure 43e), Spruce Tree Canyon Head #141 (l.o. 2011) (Figure 43e), and Cliff Palace Spring #5 (l.o. 2022) (Figure 43f) were within 10 meters of a hiking trail.

In an effort to understand the current condition of springs in MEVE, we visualized the average annual spring season discharge rates (liters/second) (Figure 44a) and average annual pool volumes (m^3) (Figure 44b) of springs from 1984 to 2023. If a measurable flow was present, surveyors measured spring discharge primarily by utilizing a volumetric method. The pool volume refers to the dimensions of a surface water pool at a spring. To ensure consistency, we only analyzed the “Pool 1” volume of each feature, as springs can vary in the number of pools they have, but the majority of entries in the dataset only had values for “Pool 1”. We chose to use this metric instead of spring area, which includes areas with soil moisture and phreatophytic vegetation (plants that depend for their water supply upon ground water that lies within reach of their roots), given differences in how observers had determined spring area (personal communication, Andrew Spear, MEVE NP Physical Science Program Manager, October 2024). While there is substantial year-to-year variation in both spring discharge and pool volume, Figure 44 does not indicate a clear trend in either metric across the period over which seeps and springs were monitored.

We assessed the past five years of springs data to understand the degree to which feral/trespass livestock are present at MEVE seeps and springs. Of only 27 seeps/springs that have been monitored at least once within the past five years within the Park, four of those (15%) had a documented trespass of a horse or cow during the most recent survey. Trespass livestock are a matter of concern in the park as grazing, even sporadically, of sensitive wetland and riparian plants and trampling of spring margins have been shown to have profound impacts to these systems, ranging from altering plant community composition to increasing turbidity of pools.

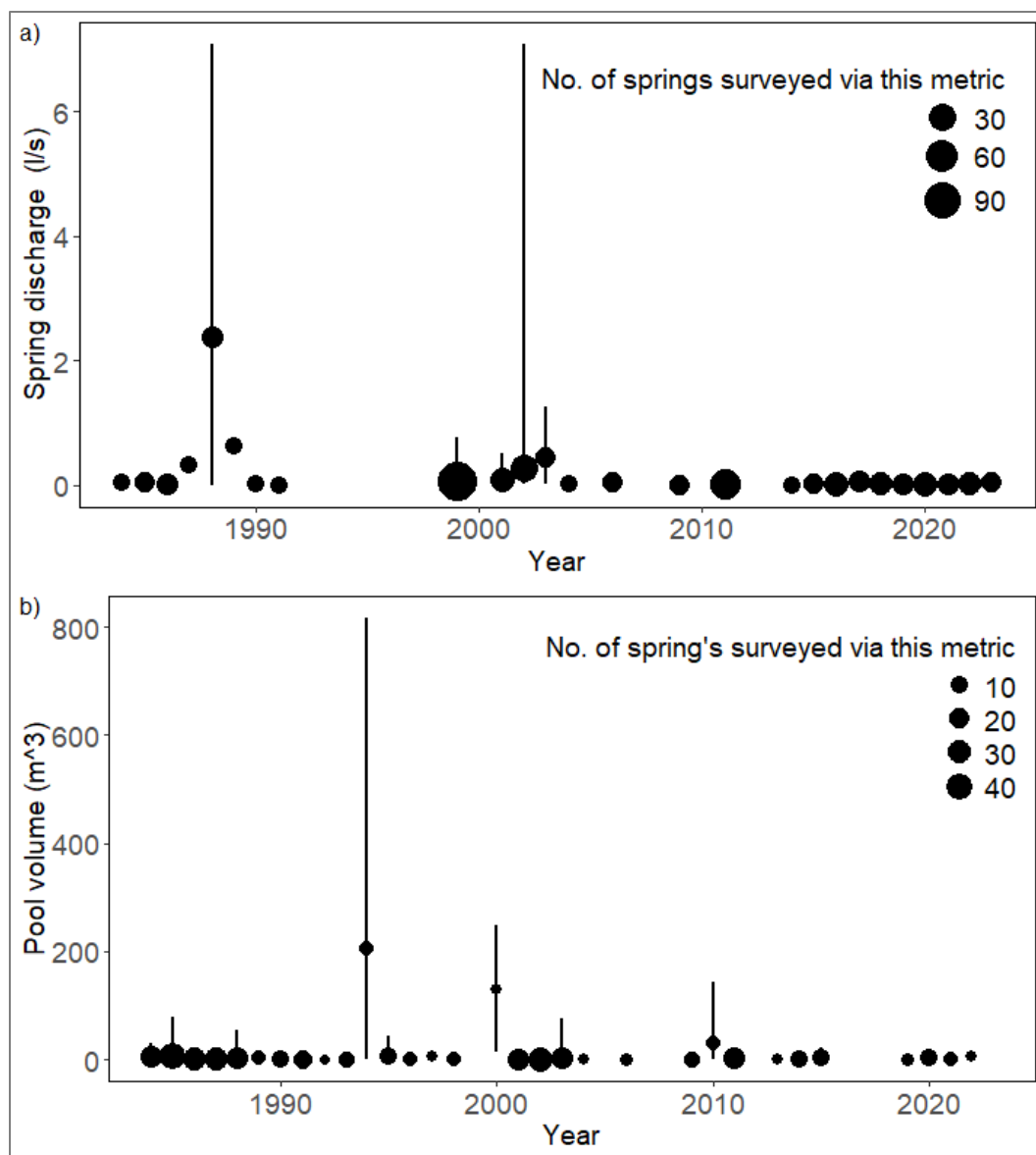


Figure 44. a) Average annual spring discharge rates (liters/second) and b) average annual pool volumes (m^3) of MEVE springs from 1984 to 2023. “MEVE springs” do include springs outside of the Park boundary that are thought to be hydrologically connected to springs in the Park (incidentally, the largest outlier in terms of spring volume in 1994 is technically outside of MEVE). The vertical lines represent the minimum and maximum values per year, and the point size represents the number of springs surveyed in a given year. Both metrics were not surveyed every year. Created using data from MEVE staff.

NPS / CONSERVATION SCIENCE PARTNERS

Chapter 4. Management Considerations

Our management considerations are grouped by the focal resources evaluated in Chapter 3 and are based on the NRCA findings and information gathered to provide managers with next steps for furthering science-informed management for a given resource. While the purpose of an NRCA is to synthesize information and deliver natural resource conditions or status of knowledge summaries for topics of Park interest, the purpose of the NPS Resource Stewardship Strategy (RSS) and other park planning processes is to establish resource goals and strategies to achieve desired resource outcomes. We offer recommendations in the sections that follow to support MEVE staff as they begin the process of developing the RSS for the Park.

4.1 Avifauna Monitoring and Management Recommendations

Passerines

Based on our findings, it is evident that the post-fire landscape at MEVE supports a dynamic range of bird species responses, particularly in areas where native deciduous vegetation has increased. To manage these post-fire habitats effectively, managers should prioritize maintaining or enhancing deciduous cover in areas where PJ woodlands may not naturally regenerate. This can be done by promoting native shrubs and trees that offer suitable habitat for the bird species that benefit from increased deciduousness, such as blue-gray gnatcatcher and Virginia's warbler. The benefits of this may be two-fold; planting species from families such as Rosaceae, which are known to resist non-native grass invasion, can provide critical habitat features, and even facilitate PJ regeneration as nurse shrubs (Redmond et al. 2014). This could be a viable strategy not only to maintain structural habitat components that are important to birds (Redmond et al. 2023), but also by promoting adaptation to climate change-mediated habitat conversion as the Southwest continues to increase in aridity (McKinnon et al. 2021).

Additionally, given that some bird species, such as green-tailed towhee and chipping sparrow, have shown vulnerability in burned areas with no significant increase with time-since-fire, it is essential to preserve structural features like tall trees and implement methods to resist further habitat conversion. If the management focus is bird species vulnerable to fire and post-fire effects, managers should be cautious of practices that may lead to a complete shift from woodland to grassland, especially in areas that have experienced high-intensity fires or are vulnerable to invasion by non-native grasses like cheatgrass (Floyd et al. 2006; Getz and Baker 2008). Maintaining a mosaic of habitat types, including refugia with existing woodland elements, can support a broader range of species and enhance ecological resilience. Measures such as creating fuel breaks, removing non-native species, and promoting native understory growth (via prescribed fires & removal of feral horses) can help safeguard these habitats from future fires and support the persistence of MEVE bird species (Coop et al. 2020; Stevens et al. 2021).

Recommendation for Future Studies to Support Management

Future research involving passerines in MEVE should investigate the proliferation of grassland species in areas that might be vulnerable to habitat conversion in MEVE and continue to track PJ bird

persistence as post-fire succession continues to unfold. Based on the literature (Erdman 1970; Huffman 2012) PJ woodland regeneration can unfold over the course of decades—particularly for piñon pines, which are slower-growing than juniper. Due to MEVE’s atypical fire activity throughout the past 50 years or so (with multiple fires occurring in the early 2000s with uncharacteristically short fire-return intervals) (Huffman 2012), the next 5–10 years may be pivotal in determining whether type conversion will take hold in MEVE or whether PJ will reestablish in burned areas. Further, the pinyon jay was excluded from our analyses due to a small sample size across datasets. This species has a mutualistic relationship with piñon pines (Ligon 1978) and is currently experiencing a steeper and more sustained population decline than any other PJ woodland songbird (Boone et al. 2018; Sauer et al. 2019). Because of the potential for fire-driven type conversion within MEVE, the vulnerability of this species, and limited data, we recommend that MEVE continues to explore options for targeted pinyon jay monitoring in the Park.

Given the relatively small size of the Park, it would also be helpful to analyze avian habitat, along with trends of key bird species, within a buffer around the Park, or even throughout a given species’ entire range. This could help to determine which habitat attributes are most important, and therefore, which should be prioritized for improvement within MEVE. Some intact PJ woodlands remain on the Ute Mountain Ute Reservation adjacent to the park, and collaboration with the UMU Tribe to compare passerine occupancy between the two areas would be an insightful starting point. Future questions to explore include: (1) have fires also affected PJ stands in nearby areas? and (2) if there is still unburned PJ habitat surrounding the Park, are PJ species within the Park not showing as strong declining trends as places losing PJ? It would also be helpful to compare occupancy rates for species by habitat type within MEVE compared to the surrounding region. Lastly, given the vegetation changes that have already occurred within MEVE due to fire, it would be informative to model predicted changes in bird occupancy or trend due to expected changes in vegetation with different climate forecasting scenarios.

4.2 Deer and Elk Monitoring and Management Recommendations

We recommend that MEVE continue their partnership with CPW and consider increased monitoring of deer and elk within the Park. Similarly, MEVE should continue its partnership with the Ute Mountain Ute Tribe, which manages wildlife within the exterior boundaries of its trust lands. Additional monitoring that would be helpful includes through surveys (e.g., by air), trail cameras, or mark-resight studies. Regardless of methods employed to better track ungulates in the Park, it would be beneficial to assess ungulate biology within the park (e.g., habitat use and selection), and whether more deer and elk move into the Park at the beginning of hunting season.

In addition, we recommended that MEVE continue efforts, including in partnership with the Ute Mountain Ute Tribe, to remove all feral/trespass livestock from the Park, given competition with deer and elk. Assuming all feral/trespass livestock are eventually removed from MEVE, the Park may want to consider increasing funding and other efforts pertaining to perimeter fence repair and maintenance to maintain livestock exclusion. The best way to achieve this may be to incorporate the boundary fencing program into the Facilities Maintenance program in partnership with the Natural Resource program, which could be a more reliable source of funds. In all cases, we recommend that

any new fencing erected be compliant with Best Management Practices for wildlife passage (i.e., height of top rail that deer and elk can jump over, and height of bottom rail that pronghorn can crawl under).

4.3 Rare Plants Monitoring and Management Recommendations

Chapin Mesa Milkvetch

We recommend the Colorado Natural Heritage Program or MEVE Park staff continue their regular CMM monitoring, with increased coordination and involvement with MEVE biologists. Future monitoring should put particular emphasis on years following average or above-average winter moisture levels when dormancy is less likely (>35% below winter average), as advocated by Rondeau (2023). When estimating total counts of CMM, we recommend factoring in dormancy rates, as Rondeau (2023) stressed their importance to more representative population estimates. Furthermore, density data should be collected across all 2001 baseline transects to provide a more comprehensive understanding of CMM population estimates (Rondeau 2023). Rondeau (2023) further recommended collecting species composition data (trees, shrubs, grasses, forbs, bare ground/litter/crust) from each transect in both burned and unburned areas to compare the difference between plots with increasing and decreasing CMM density trends.

In order to protect park infrastructure, MEVE has installed fuel breaks by clearing and thinning PJ woodlands 100 feet from existing buildings and along the Ute Mountain Ute Reservation boundary. Although fuel treatments may negatively impact CMM found close to infrastructure, they should help to reduce the extent of stand replacing fires in CMM habitat, ultimately benefiting this species (Mesa Verde National Park 2018). Thus, we recommend that where possible, if the Park's goal is to promote CMM populations, they continue to minimize the intensity of proposed fuel management treatments in CMM habitat to reduce negative impacts while still maintaining the benefits of preventing wildfire spread, as established in the CMM Conservation Plan (Mesa Verde National Park 2018). In general, we recommend that MEVE continue to follow the management recommendations contained in the Conservation Strategy starting on page 24 of the CMM Conservation Plan (Mesa Verde National Park 2018).

Future CMM studies should examine how increased competition with herbaceous species, as well as site-level species composition, soil texture, soil moisture, and soil temperature correlate with CMM vital rates and population trends (Rondeau 2019; Rondeau et al. 2022). Such information could provide guidance to managers in restoring CMM in burned areas in the future. Additionally, studies examining correlations between altered light availability (i.e., with missing PJ canopy after fire), increased temperature, insolation, reduced pollinator abundance, and changes in herbivory rates with CMM population trends could provide additional information on causes of CMM declines, as suggested by Rondeau et al. (2022).

In the conclusions of Rondeau et al. (2022), the authors underscored the extirpation risk of CMM in burned areas of formerly intact PJ, likely due to the establishment of novel grassland communities catalyzed by non-native and/or invasive grass expansion. As such, Rondeau et al. (2022) placed a

high urgency on preserving remaining old-growth PJ woodland in MEVE¹³. Furthermore, the authors recommended expanded research on post-fire vegetation management in the Park, especially as it relates to assisting succession of plant communities and reducing invasive grasses like western wheatgrass. Rondeau (2019) further recommended the development of a post-fire seed mix of ~50 species, containing forbs, shrubs, limited grasses (only bunch grasses and not western wheatgrass), and CMM to aid in post-fire recovery and reduce establishment of non-native grasses.

Cliff Palace Milkvetch

CPM, endemic to the Mesa Verde cuesta, depends on seed production and the seed bank to maintain a stable and viable population. With the current low numbers of plants known to exist (Stowell 2022a), seed production is likely limited, thus contributing to continued decline. Monitoring of this species, including the seed bank, should continue and be expanded. While there was no monitoring of CPM in MEVE in 2023 or 2024, the Park has experienced higher precipitation in years since 2022. Continued monitoring would allow the park to determine if CPM populations have rebounded in response to increased precipitation.

For future CPM monitoring, MEVE staff should continue to monitor the Moore (2004–2007) plots, while also expanding monitoring to additional locations and transects to gain a more robust understanding of the species across the park. Data collection should include annual fruit, flowering and seed production. A seed bank study, seed viability, moisture germination and recruitment requirements (timing and amount), and pollinator studies (since it is an early spring species subject to climate change) are also recommended for this species. Stowell (2022a) also suggested that larger plots than were used in the 2021 and 2022 monitoring be established that capture more CPM plants, since the areas where the original (2004–2007) Windom Floristics plots were located no longer support the number of plants needed to generate enough meaningful data to track long term trends (Long Mesa plots are the exception). Even when the plot size was expanded in the 2022 surveys, this still did not capture a sufficient number of individual plants to have a large enough sample for making observations about population-level trends.

The recent, major decline of CPM warrants immediate management action if the Park does not want this species to become extirpated from MEVE. Documented threats to CPM include trampling by visitors on trails, impacts by Park infrastructure (roads, waterlines), erosion, non-native species and climate change. Additional investigation is needed to determine how to address these threats to the species to ensure persistence in MEVE.

Recommendation for Future Studies and Monitoring (All Species) to Support Management

The Park should continue its partnership with the Ute Mountain Ute Tribe and conduct additional joint surveys of rare plants that span the boundary of MEVE and the Reservation. The Park should also consider developing and implementing a monitoring plan and protocol for the rare species that are most vulnerable in MEVE. In addition to CMM and CPM, which are being monitored, other

¹³ This is also a key recommendation included in the Conservation Strategy in the CMM Conservation Plan (Mesa Verde National Park 2018).

vulnerable species include *Hackelia gracilenta* and *Townsendia glabella*. For the other species a “re-visit” (every 2–3 years) survey protocol of a representative sample of occurrences should be developed to ensure species are extant on the landscape. The Park could also consider a partnership with a botanical garden for ex-situ conservation of some of its most vulnerable, endemic species.

Several species are vulnerable to park activities and visitation impacts; these include trampling of *Collomia grandiflora* at the Geologic Overlook and the occurrences of CPM along the Mug House and Spring House trails. Localized stress occurs to CMM, CPM, *Hackelia gracilenta*, and *Aletes mcdougalli* ssp. *breviradiatus* from trampling along most of the trails and roads on Chapin and Long Mesas. The only documented occurrence of *Townsendia glabella* has an administrative road and water line running through its population. Documenting impacts from visitation, Park infrastructure, and roads and road maintenance to rare plants should be considered in all monitoring and survey protocols. If it is discovered that these activities are having measurable impacts, the Park should consider closing or redirecting the trails causing the worst impacts or employing alternative mitigation efforts.

4.4 Mancos River Monitoring and Management Recommendations

We recommend that SCPN continue to monitor water quality and macroinvertebrates within MEVE and provide a summary of the analysis in the next round of status and trends reports to the Park. Monitoring should include annual water quality and discharge measurements and macroinvertebrate sample collections from both of the existing SCPN data collection sites within the Park (MAN 01 and MAN 02). We also recognize that SCPN may need to establish a new, third monitoring station for macroinvertebrates that has less chance of being impacted by stream restoration activities (such as placement of Beaver Dam Analogues and Post-assisted Log Structures).

We recommend that MEVE continue to actively manage MEVE’s section of the Mancos River corridor and associated riparian zone in order to keep it as structurally complex, healthy, and extensive as possible, so it will continue to be an attractive stop-over habitat to neotropical migratory birds, even in a changing climate. The MEVE stretch of the Mancos River could be part of an important dispersal and/or migration corridor between the Chuska and San Juan mountains for many different taxa. It is the only north-south river corridor that is not heavily impacted by human use in this region. With projected climate changes, these relatively undisturbed corridors may become increasingly important to conserve and protect, as wildlife seek to move within climate refugia.

To this end, MEVE may want to consider continued partnership with the Mancos Watershed Group to explore the use of Post-assisted Log Structures and other Low-tech Process-based Restoration structures in places where hydrologic restoration is desired on the MEVE river segment. Additionally, this stretch of the river may be a good place to consider ways to naturally recruit or augment current beaver populations. The artificial structures recently placed in the river corridor seem to so far be achieving intended purposes and might help attract beavers from outside the Park.

Because the Park’s stretch of the Mancos River is not grazed by livestock, this presents an opportunity to continue with efforts to remedy past channelization and reconnect the river with its

historic floodplain, without livestock grazing removing streamside vegetation (including cottonwood seedlings) and contributing to bank erosion. Restoration efforts aimed at channel aggradation could have long-term pay-offs including increased soil moisture and a restored floodplain, with potential benefits for species like New Mexico meadow jumping mouse, and even yellow-billed cuckoo, as increased cottonwood seedlings and saplings eventually recruit into the canopy.

For native and non-native fish monitoring and management, MEVE should continue to engage with both Colorado Parks and Wildlife and the Ute Mountain Ute Tribe on future native fish stocking into the Mancos River. If possible, fish surveys, in partnership with CPW and regional NPS fish biologists, should be conducted annually. We note that continuing with active stream restoration projects, such as BDAs and PALs (which ideally will continue to be maintained and rebuilt by beavers if washed out in spring run-off), should ultimately have a positive effect on native fish habitat. While PALs and natural beaver dams can create ponds that, at least temporarily, inundate spawning beds, these structures also create complexity and introduce conditions with which native species evolved. Moreover, the new pools created by dams and dam-like structures can provide refuge habitat for fish when summer flows decline or even disappear completely. We recommend that the Park engage with the Mancos Watershed Group and Mancos Conservation District to conduct RSRA surveys within the Park every ~3 years. The fish/aquatic habitat category of the protocol will help assess whether native fish habitat continues to improve along with active stream restoration projects within MEVE.

Lastly, we recommend that MEVE continue to engage with the Mancos Conservation District, MWG, and other local stakeholders as they complete the Mancos Watershed Stream Management Plan in the coming year. This plan will map a consensus-based future path for management of the Mancos River from its headwaters to the confluence with the San Juan River, including a holistic assessment of river health, identification of threats¹⁴ and resource concerns, future desired conditions, opportunities for collaboration, and recommended actions. The Park's involvement in this process will be key.

Recommendation for Future Studies to Support Management

As recommended by Stumpf (2024), and as the effects of climate change increase over the coming decades, future monitoring should include indicators of thermal and hydrologic stress to better understand community responses to those stresses as well as their effects on the outcome of management actions. Moreover, as the NPS prepares to develop the RSS for the Park, we recommend that they work with NPS's Climate Change Response Program to develop a correlative analysis of climate change and the Mancos River hydrograph over time, with future projections of river flow.

¹⁴ For example, if there are extensive stands of Russian olive above the Park, then monitoring should be done as it can spread quickly in the presence of major flooding events.

4.5 Upland Vegetation Monitoring and Management Recommendations

Piñon-juniper Woodland

Starting in 2019, MEVE staff examined the effects of mechanical treatments performed in PJ woodlands in MEVE in order to reduce fire risk. Staff examined effects on biodiversity, the recruitment of PJ, spread of non-native plants, and the presence of the Chapin Mesa milkvetch (Spector et al. 2019). Phillips et al. (2023) discussed the effects of thinning treatments within burned and intact MEVE PJ stands. Thinning methods on Long Mesa significantly altered the composition of the plant community, and yet the desired goal of increasing native grass cover was not achieved (Phillips et al. 2023). Phillips et al. (2023) concluded that the implications of vegetation thinning are nuanced; while this tactic has the potential to resist habitat type conversion by decreasing the likelihood of future fire, it also remains unclear how to minimize unintended ecological consequences. As such, we recommend the continuation of this study to further understand the long-term effects of thinning in PJ communities.

If the management goal is the persistence of PJ woodlands with native understories, we do not recommend further mechanical PJ thinning treatments¹⁵ in MEVE at this time. This recommendation was reiterated by Swan (2024), who stated in the recent SCPN PJ monitoring report for MEVE: “In old-growth ... persistent woodlands like those found at ... MEVE, stand thinning is not recommended as it is unlikely to increase drought resiliency and risks introducing non-native annual grasses.” The threat of mechanical thinning treatments introducing non-native grass species onto a site was also echoed by Shinneman et al. (2023). Additionally, we highlight the value of the Park as a unique research environment in which to witness passive restoration after severe drought, fire, or other natural disturbance. Modeling climate change impacts on the population demographics of PJ, Noel et al. (2023) predicted that future conditions within this part of the Colorado Plateau will support PJ, and that MEVE might even be a climate refugia for this habitat type. Further, reframing future research objectives to incorporate the entire Mesa Verde cuesta could present opportunities to compare long-term, passive restoration strategies¹⁶ within the Park to more active post-disturbance management outside of the Park—especially considering the unburned, intact PJ woodland refugia that occurs just outside the Park boundary as well as inside MEVE.

In her recent report summarizing long-term vegetation plot data collected by SCPN on intact PJ woodlands in MEVE, Swan (2024) commented on the good conditions found in SCPN monitoring plots, noting very low incidence of cheatgrass. Swan underscored that to keep these systems relatively free of cheatgrass, there needs to be limited exposure to other disturbance types, as soils within undisturbed PJ woodland stands have been found to be notably resistant to weed invasion (Floyd et al. 2006). In particular, trespass livestock grazing can generate ground disturbance and introduce nonnative seeds, while selectively grazing certain species can lead to establishment and spread of non-native species (Fleischner 1994; Floyd et al. 2003; Swan 2024). As mentioned above,

¹⁵ By “mechanical treatment” we are not referring to a few, selected (including dead) trees cut by hand and removed from the site. The literature (including Phillips et al. 2023) indicates this practice is relatively benign, and is typically on a different scale than typical, larger mechanical vegetation treatments.

¹⁶ We do recommend, however, fighting large fires that may threaten intact PJ stands—see below.

we recommend MEVE continue monitoring the effects of trespass livestock, and to continue trespass livestock removal efforts, in PJ communities and across the Park.

Finally, as mentioned earlier in the report, the Park's Douglas fir woodlands have been severely impacted by fire activity and are not likely to recover (personal communication, Tova Spector, USFS Intermountain Region 4, October 2024). However, the extent of this loss has not been quantified, resulting in both data and knowledge gaps. As such, we recommend that the Park initiates research that aims to address these gaps, with targeted and extensive monitoring within the few remaining stands in the Park.

Non-native Species

Our analyses above indicate that, while non-native annual grasses and forbs are driving MEVE upland vegetation system structure during and immediately post-fire, it is not driving structure or function in the post-fire (long-term) period. Still, we recommend that the Park continue with its efforts to monitor current outbreaks of invasive, non-native plants and work to prevent new ones. The volume of herbicide applied to non-native plant outbreaks in the future should be more carefully and consistently documented to evaluate the success of MEVE's non-native plant program and inform future chemical application efforts. The most important recommendation we can make to MEVE regarding non-native plants is to increase monitoring and documentation of whether prior treatment efforts have been effective, including mapping the aerial extent of infested areas one year post treatment, whether the species of concern is alive or dead within the area, and general estimation of non-native plant density within those areas.

We recommend that MEVE continue to use its toolbox of mechanical, cultural, chemical, and biological means to treat non-native plants (National Park Service 2015), including bagging and removing all seed heads from mechanically treated areas wherever possible (Knight et al. 2021). The Park should continue its program of native seed collection, including rare and endemic plant seeds, for use in revegetation projects (Knight et al. 2021), and should consider planting native shrubs in areas most vulnerable to post-fire weed invasion, partly because of their role as nurse plants for piñon seedlings (Floyd et al. 2006). In addition, MEVE may want to consider halting use of the Bio-control bacteria MB906 for cheatgrass suppression, as recent research has found that the bacteria has an inconsistent or no effect on plant communities (Knight et al. 2021). However, the use of other biological controls such as Canada thistle gall fly (*Urophthora cardui*), which was used following the Chapin 5 fire of 1999, may be beneficial in some areas. In general, we suggest that the best technique for managing non-native species, especially cheatgrass, is to limit soil disturbance in areas that are not yet affected. Managing cheatgrass in the Park has proven to be difficult (e.g., the accidental introduction of cheatgrass, which also germinates faster than natives, in post-fire reseeding efforts (Copeland et al. 2019; Rondeau et al. 2022)). As such, we recommend minimizing soil and other disturbances in intact systems to resist post-disturbance non-native species colonization.

We recommend that MEVE complete the 2015 Draft Invasive Plant Management Plan (IPMP), and continue implementing the goals, priorities and strategies laid out and thus far generally adhered to in the EA, including using the decision-making trees developed in the EA and the action triggers

recommended in the EA for weed prevention and early detection (National Park Service 2015). The Park should also continue to implement a comprehensive monitoring program to better understand trends in priority non-native plant infestations, non-target treatment impacts, and treatment efficacy (National Park Service 2015), as outlined in the 2014 IPMP EA.

In summary, we recommend that MEVE continue to give high priority for control of non-native plant species or occurrences that:

1. May adversely affect or threaten natural communities, wildlife habitat, or rare plant species.
2. Occur within the historic zone or the Research Natural Area on Park Mesa in MEVE.
3. Occur in developed areas that are “hot spots” or pathways for infestations to spread.
4. Threaten the integrity of a cultural landscape.
5. Are listed by the state and/or county as high priority for eradication or control.
6. Occur in areas where rapid dispersal to other areas of MEVE is possible, or near the Park boundary, posing a threat of spreading to neighboring lands.
7. Are infestations of new non-native plant species, having never occurred in MEVE before.
8. Species that have a high potential for ecological impact calculated in a Risk Assessment.

Fire, and Post-Fire Monitoring

Despite the importance of fire to this ecotype, if the Park’s goal is to preserve the few remaining intact, old growth PJ stands, we recommend the continued, targeted suppression of fires that threaten these stands as old growth PJ is generally unlikely to recover from major fires.

The Park should continue Long Term Post Fire monitoring (LTPFM) to further understand the ongoing development of vegetation structure in burned sites. If MEVE is interested in whether these landscapes are experiencing type conversion, additional long-term data collection must occur. Beyond data regarding vegetation structure, we recommend collecting other climatic and soil-related data from burned areas, including soil moisture, biocrust composition (as seen in earlier LTPFM reports) and moderate temperature, as documented by Phillips et al. (2023), given the authors’ insistence that these factors will impact PJ woodland regeneration. Furthermore, future monitoring efforts may be improved by adding new plots in burned unseeded areas of the Park with similar characteristics to assess the effectiveness of seeding efforts (Spector et al. 2019), which we fully support.

We also recommend regular literature reviews to evaluate the most up-to-date post-fire seeding recommendations as more recently, researchers have observed that seeding grasses after vegetation removal led to increasing non-native species richness and declining native richness (Copeland et al. 2019). Grasses also often compete with shrub seedlings, which could decrease shrub establishment (Yelenik and Levine 2010). This is problematic given that shrubs are important nurse plants to PJ seedlings, and PJ regeneration in burned areas may therefore hinge on the presence of these shrubs (Floyd et al. 2021). Based on recent research, MEVE may consider seeding a post-fire mix consisting

of forbs, shrubs, and limited bunchgrasses (i.e., no western wheatgrass which, as we report above, has been implicated in low recovery rates post fire).

4.6 Seeps and Springs Monitoring and Management Recommendations

We recommend that MEVE continue to monitor springs with their current assessment method, which is fairly new. Because the current springs dataset combined findings from two or three different assessment methods over time, variables and indicators and measures recorded for them were often not consistent in the dataset. This, combined with sometimes a decade or more between repeat assessment visits to some springs, makes it difficult to perform time series analysis with the dataset. We understand that resources are lacking and so not every spring can be visited and assessed every year. However, if the majority of springs that are known to be flowing can be assessed every second or third year with the new protocol, this would enable more robust analysis for the next spring condition assessment in the Park. Alternatively, if springs are not being visited because of either lack of resources or because those springs no longer flow, this information would be helpful to add to the dataset. Additionally, pending findings from a more robust sampling effort, if trespass livestock are shown to threaten the health or integrity of Park springs, MEVE could consider either fencing around springs could be considered, or utilizing Zeedyk rock structures to retain moisture and to help bring wetland and native vegetation back to springs that have been heavily disturbed.

Recommendation for Future Studies to Support Management

We recommend that MEVE work in conjunction with the NPS's Climate Change Response Program and Water Resources Division to explore the possibility of applying a water balance model to estimate changes in hydrologic conditions that may affect MEVE seeps, springs, and other resources under different climate futures. This exercise could be helpful to MEVE as work begins on the Park's RSS.

Literature Cited

- Allred, B. W., B. T. Bestelmeyer, C. S. Boyd, C. Brown, K. W. Davies, M. C. Duniway, L. M. Ellsworth, T. A. Erickson, S. D. Fuhlendorf, T. V. Griffiths, V. Jansen, M. O. Jones, J. Karl, A. Knight, J. D. Maestas, J. J. Maynard, S. E. McCord, D. E. Naugle, H. D. Starns, and D. R. Uden. 2021. Improving Landsat predictions of rangeland fractional cover with multitask learning and uncertainty. *Methods in Ecology and Evolution*, 12(5), 841–849. <https://doi.org/10.1111/2041-210X.13564>
- Amundson, C. L., J. A. Royle, and C. M. Handel. 2014. A hierarchical model combining distance sampling and time removal to estimate detection probability during avian point counts. *The Auk: Ornithological Advances*, 131(4).
- Anderson, D. G. 2004. Population Status Survey of Schmoll's Milkvetch (*Astragalus schmollii* C.L. Porter): Final Report. Colorado Natural Heritage Program, Colorado State University.
- Anderson, M., S. A. Goeking, J. Gragg, S. A. Green, A. Jones, M. O'Brien, and D. Tart. 2024. Guidelines for Riparian Forest Restoration in Utah [USFS-RMRS Gen Tech Report]. U.S Forest Service.
- Andrews, C., J. Bradford, J. Norris, J. Gremer, M. Duniway, S. Munson, L. Thomas, and M. Swan. 2019. Describing Past and Future Soil Moisture in the Loamy Mesa-Top Pinyon-Juniper Woodland Ecosystem in Mesa Verde National Park. (NPS/CHCU/NRR—2019/1998.). National Park Service.
- Balch, J. K., B. A. Bradley, C. M. D'Antonio, and J. Gómez-Dans. 2013. Introduced annual grass increases regional fire activity across the arid western USA (1980–2009). *Global Change Biology*, 19(1), 173–183. <https://doi.org/10.1111/gcb.12046>
- Barrett, G. W., G. M. van Dyne, and E. P. Odum. 1976. Stress Ecology. *BioScience*, 26(3), 192–194. <https://doi.org/10.2307/1297248>
- Barton, A. M., and H. M. Poulos. 2018. Pine vs. oaks revisited: Conversion of Madrean pine-oak forest to oak shrubland after high-severity wildfire in the Sky Islands of Arizona. *Forest Ecology and Management*, 414, 28–40.
- Bates, J. D., R. F. Miller, and T. Svejcar. 2007. Long-term vegetation dynamics in a cut western juniper woodland. *Western North American Naturalist*, 67(4), 549–561.
- Belnap, J., and D. A. Gillette. 1997. Disturbance of biological soil crusts: Impacts on potential wind erodibility of sandy desert soils in southeastern Utah. *Land Degradation & Development*, 8(4), 355–362. [https://doi.org/10.1002/\(SICI\)1099-145X\(199712\)8:4<355::AID-LDR266>3.0.CO;2-H](https://doi.org/10.1002/(SICI)1099-145X(199712)8:4<355::AID-LDR266>3.0.CO;2-H)

- Belsky, A. J., and J. L. Gelbard. 2000. Livestock grazing and weed invasions in the arid West. Oregon Natural Desert Association Portland. <https://www.alamedacreek.org/take-action/pdf/Belsky%20and%20Gelbard%202000.pdf>
- Bennet, S., J. Wheaton, N. Bouwes, S. Shahverdian, W. W. MacFarlane, and E. Portugal. 2019. Planning for Low-Tech Process-Based Restoration (Chapter of Book DOI: 10.13140/RG.2.2.15815.75680).
- Bird Conservancy of the Rockies. 2024. Bird Conservancy of the Rockies. Bird Conservancy of the Rockies. <https://www.birdconservancy.org/>
- Boone, J. D., E. Ammon, and K. Johnson. 2018. Long-term declines in the Pinyon Jay and management implications for piñon–juniper woodlands. Trends and Traditions: Avifaunal Change in Western North America. Studies of Western Birds, 3, 190–197.
- Bradford, J. B., D. R. Schlaepfer, W. K. Lauenroth, and K. A. Palmquist. 2020. Robust ecological drought projections for drylands in the 21st century. Global Change Biology, 26(7), 3906–3919. <https://doi.org/10.1111/gcb.15075>
- Bradley, B. A., C. A. Curtis, E. J. Fusco, J. T. Abatzoglou, J. K. Balch, S. Dadashi, and M. -N. Tuanmu. 2018. Cheatgrass (*Bromus tectorum*) distribution in the intermountain Western United States and its relationship to fire frequency, seasonality, and ignitions. Biological Invasions, 20(6), 1493–1506. <https://doi.org/10.1007/s10530-017-1641-8>
- Brasher, A. M., C. M. Albano, R. M. Close, L. M. Freeman, S. E. Monroe, A. E. Stumpf, C. Snyder, and L. P. Thomas. 2011. Aquatic macroinvertebrate monitoring protocol for the Southern Colorado Plateau Network (Natural Resource Report NPS/SCPN/NRR—2011/460.). National Park Service.
- Breshears, D. D., N. S. Cobb, P. M. Rich, K. P. Price, C. D. Allen, R. G. Balice, W. H. Romme, J. H. Kastens, M. L. Floyd, and J. Belnap. 2005. Regional vegetation die-off in response to global-change-type drought. Proceedings of the National Academy of Sciences, 102(42), 15144–15148.
- Brown, N. 2021. Mesa Verde National Park 2020 Wildlife Program Report (p. 31) [Annual Wildlife Program Report]. Mesa Verde National Park.
- Brown, K., and S. Roberts. 2021. Mine Reclamation in the East Mancos River / Thunder Mine [PowerPoint].
- Buckland, S. T., D. R. Anderson, K. P. Burnham, J. L. Laake, D. L. Borchers, and L. Thomas. 2001. Introduction to distance sampling: Estimating abundance of biological populations. Oxford university press. <https://academic.oup.com/book/53746>
- Climate Change Response Program. 2024. Mesa Verde National Park climate change exposure summary.

- Clinton, B. D. 2011. Stream water responses to timber harvest: Riparian buffer width effectiveness. *Forest Ecology and Management*, 261(6), 979–988. <https://doi.org/10.1016/j.foreco.2010.12.012>
- Colorado Department of Public Health and Environment. 2003. Standard operating procedures for collecting and preserving benthic macroinvertebrate samples. Colorado Department of Public Health and Environment, Water Quality Control Division-Monitoring Unit.
- Colorado Department of Public Health and Environment, Water Quality Control Commission. n.d. Regulation No. 34—Classifications and Numeric Standards for San Juan River and Dolores River Basins. (Colorado Code of Regulations 5 CCR 1002–34).
- Colorado Natural Heritage Program. 2023. Biodiversity Tracking and Conservation System. (Data exported October 2023) [GIS data]. Colorado State University.
- Colorado Parks and Wildlife. n.d.. Rocky Mountain Elk. Colorado Parks and Wildlife. Retrieved March 4, 2024, from <https://cpw.state.co.us/species/elk>
- Colorado Parks and Wildlife. 2015. State Wildlife Action Plan (p. 865). Colorado Parks and Wildlife.
- Colorado Parks and Wildlife. 2017. Colorado’s Mule Deer Story. Colorado Parks and Wildlife. <https://cpw.state.co.us>
- Colorado Parks and Wildlife. 2022. Integrated Water Quality Monitoring and Assessment Report.
- Colorado Parks and Wildlife. 2023. Aquatics Data Management System (ADAMAS) [Dataset]. Colorado Parks & Wildlife. <https://form.asana.com/?k=nBAmn36btfY-ifvb0S0JPw&d=5295706742479>
- Colorado Parks and Wildlife. 2024a. Colorado Hunting Atlas. Colorado Hunting Atlas. <https://ndismaps.nrel.colostate.edu/index.html?app=HuntingAtlas>
- Colorado Parks and Wildlife. 2024b. Deer-Statistics. Deer-Statistics. <https://cpw.state.co.us/hunting/big-game/deer/statistics>
- Colorado Parks and Wildlife. 2024c. Elk—Statistics. Elk - Statistics. <https://cpw.state.co.us/hunting/big-game/elk/statistics>
- Colorado Rare Plant Guide. 2024. *Townsendia glabella*. *Townsendia Glabella*. <https://cnhp.colostate.edu/rareplant/details/?plantID=20834>
- Colorado Water Conservation Board. 2024. Public Information: <https://cwcb.colorado.gov/public-information>
- Colyer, M. 2006. History and Nature along the Mancos River Corridor [MEVE Internal report]. National Park Service.

- Colyer, M., and G. SanMiguel. 2001. FY 2000 Annual Report: Natural Resources [Mesa Verde National Park].
- Cooley, C. P., A. Holland, M. Cowardin, M. Flenner, T. Balzer, J. Stiver, E. Slezak, B. Marette, D. Neumann, and T. Elm. 2020. Big Game Winter Range And Migration Corridors. Colorado Parks and Wildlife.
- Coop, J. D., S. A. Parks, C. S. Stevens-Rumann, S. D. Crausbay, P. E. Higuera, M. D. Hurteau, A. Tepley, E. Whitman, T. Assal, and B. M. Collins. 2020. Wildfire-driven forest conversion in western North American landscapes. *BioScience*, 70(8), 659–673.
- Copeland, S. M., S. M. Munson, J. B. Bradford, B. J. Butterfield, and K. L. Gunnell. 2019. Long-term plant community trajectories suggest divergent responses of native and non-native perennials and annuals to vegetation removal and seeding treatments. *Restoration Ecology*, 27(4), 821–831. <https://doi.org/10.1111/rec.12928>
- Crafford, R. E., I. K. Breckheimer, H. J. Jones, and J. D. Coop. In preparation. Leveraging Remote Sensing to Model Bird-Habitat Associations Across Fire-Affected Piñon-Juniper Landscapes.
- Craig, G. R., and J. H. Enderson. 2004. Peregrine Falcon Biology and management.
- DeCoster, J., C. Lauver, M. Miller, J. R. Norris, A. Snyder, M. Swan, L. Thomas, and D. Witwicki. 2012. Integrated upland monitoring protocol for the Southern Colorado Plateau Network.
- Dios Miranda, J. de, F. M. Padilla, R. Lázaro, and F. I. Pugnaire. 2009. Do changes in rainfall patterns affect semiarid annual plant communities? [Do changes in rainfall patterns affect semiarid annual plant communities? - Miranda - 2009 - Journal of Vegetation Science - Wiley Online Library](https://onlinelibrary.wiley.com/doi/10.1111/j.1365-3113.2009.00411.x)
- Dunk, J. R., B. Woodbridge, T. M. Lickfett, G. Bedrosian, B. R. Noon, D. W. LaPlante, J. L. Brown, and J. D. Tack. 2019. Modeling spatial variation in density of golden eagle nest sites in the western United States. *PLOS ONE*, 14(9), e0223143. <https://doi.org/10.1371/journal.pone.0223143>
- eBird. 2024. Golden Eagle Status & Trends. Golden Eagle Status & Trends. <https://science.ebird.org/en/status-and-trends/species/goleag/trends-map>
- Eckert, S. L., and T. Clark. 2009. The Ritual Importance of Birds in 14th-century Central New Mexico. <https://journals.sagepub.com/doi/full/10.2993/0278-0771-29.1.8>
- Erdman, J. A. 1970. Pinyon-juniper succession after natural fires on residual soils of Mesa Verde, Colorado. *Brigham Young University Science Bulletin, Biological Series*, 11(2).

- Farnsworth, G. L., K. H. Pollock, J. D. Nichols, T. R. Simons, J. E. Hines, and J. R. Sauer. 2002. A removal model for estimating detection probabilities from point-count surveys. *The Auk*, 119(2), 414–425.
- Fensham, R. J., W. F. Ponder, V. Souza, and L. E. Stevens. 2023. Extraordinary concentrations of local endemism associated with arid-land springs. *Frontiers in Environmental Science*, 11. <https://www.frontiersin.org/articles/10.3389/fenvs.2023.1143378>
- Ferg, A. 2007. Birds in the Southwest. *Archaeology Southwest*. <https://www.archaeologysouthwest.org/product/asw21-01/>
- Ferrenberg, S., A. M. Faist, A. Howell, and S. C. Reed. 2018. Biocrusts enhance soil fertility and *Bromus tectorum* growth, and interact with warming to influence germination. *Plant and Soil*, 429(1), 77–90. <https://doi.org/10.1007/s11104-017-3525-1>
- Fink, D., T. Auer, A. Johnston, M. Strimas-Mackey, S. Ligocki, and O. Robinson. 2023. 2023. eBird Status and Trends, Data Version: 2022 (Version 2022) [Dataset]. Cornell Lab of Ornithology, Ithaca, New York.
- Flake, S. W., and P. J. Weisberg. 2021. Drought alters the understory of pinyon-juniper woodlands indirectly through tree dieback. *Rangeland Ecology & Management*, 76, 118–128.
- Fleischner, T. L. 1994. Ecological Costs of Livestock Grazing in Western North America. *Conservation Biology*, 8(3), 629–644. <https://doi.org/10.1046/j.1523-1739.1994.08030629.x>
- Floyd, M. L. 2003. Ancient piñon-juniper woodlands: A natural history of Mesa Verde country. University Press of Colorado.
- Floyd, M. L., M. Clifford, N. S. Cobb, D. Hanna, R. Delph, P. Ford, and D. Turner. 2009. Relationship of stand characteristics to drought-induced mortality in three Southwestern piñon–juniper woodlands. *Ecological Applications*, 19(5), 1223–1230. <https://doi.org/10.1890/08-1265.1>
- Floyd, M. L., M. Colyer, D. D. Hanna, and W. H. Romme. 2003. Gnarly old trees: Canopy characteristics of old-growth pinon–juniper woodlands. In *Ancient pinon–juniper woodlands*. (pp. 11–30). University Press of Colorado.
- Floyd, M. L., D. D. Hanna, and W. H. Romme. 2004. Historical and recent fire regimes in Piñon–Juniper woodlands on Mesa Verde, Colorado, USA. *Forest Ecology and Management*, 198(1–3), 269–289. <https://doi.org/10.1016/j.foreco.2004.04.006>
- Floyd, M. L., D. D. Hanna, W. H. Romme, and T. E. Crews. 2006. Predicting and mitigating weed invasions to restore natural post-fire succession in Mesa Verde National Park, Colorado, USA. *International Journal of Wildland Fire*, 15, 247–259.

- Floyd, M. L., W. H. Romme, and D. D. Hanna. 2000a. Fire History and Vegetation Pattern in Mesa Verde National Park, Colorado, USA. *Ecological Applications*, 10(6), 1666–1680.
<https://doi.org/10.2307/2641230>
- Floyd, M. L., W. H. Romme, and D. D. Hanna. 2000b. Fire History and Vegetation Pattern in Mesa Verde National Park, Colorado, USA. *Ecological Applications*, 10(6), 1666–1680.
<https://doi.org/10.2307/2641230>
- Floyd, M. L., W. H. Romme, and D. D. Hanna. 2021. Effects of Recent Wildfires in Piñon-Juniper Woodlands of Mesa Verde National Park, Colorado, USA. *Natural Areas Journal*, 41.
<https://doi.org/10.3375/043.041.0105>
- Floyd, M. L., W. H. Romme, M. E. Rocca, D. P. Hanna, and D. D. Hanna. 2015. Structural and regenerative changes in old-growth piñon–juniper woodlands following drought-induced mortality. *Forest Ecology and Management*, 341, 18–29.
- Friedlander, J. 1980. The status of rare and endangered plant species in Mesa Verde National Park, Colorado. Department of Range Science, Colorado State University.
- Fusco, E., J. Finn, J. Balch, R. C. Nagy and B. Bradley. 2019. Source data for “Invasive grasses increase fire occurrence and frequency across U.S. ecoregions.” Data and Datasets.
<https://doi.org/10.7275/ndsz-eh64>
- Ganey, J. L., W. M. Block, J. Dwyer, B. E. Strohmeier, and J. S. Jennes. 1998. Dispersal movements and survival rates of juvenile Mexican spotted owls in northern Arizona. *Wilson Bulletin*, 110(2), 206–217.
- Garmoe, M., and H. Perry. 2014. Long-term post-burn monitoring of former Pinyon-Juniper Woodlands at Mesa Verde National Park. National Park Service.
- Gelman, A., and D. B. Rubin. 1992. Inference from iterative simulation using multiple sequences. *Statistical Science*, 7(4), 457–472.
- Getz, H. L., and W. L. Baker. 2008. Initial invasion of cheatgrass (*Bromus tectorum*) into burned pinon-juniper woodlands in western Colorado. *The American Midland Naturalist*, 159(2), 489–497.
- Gorelick, N., M. Hancher, M. Dixon, S. Ilyushchenko, D. Thau, and R. Moore. 2017. Google Earth Engine: Planetary-scale geospatial analysis for everyone. *Remote Sensing of Environment*, 202, 18–27.
- Graeber, D., T. M. Jensen, J. J. Rasmussen, T. Riis, P. Wiberg-Larsen, and A. Baattrup-Pedersen. 2017. Multiple stress response of lowland stream benthic macroinvertebrates depends on habitat type. *Science of the Total Environment*, 599, 1517–1523.

- Guiterman, C. H., R. M. Gregg, L. A. E. Marshall, J. J. Beckmann, P. J. Van Mantgem, D. A. Falk, J. E. Keeley, A. C. Caprio, J. D. Coop, P. J. Fornwalt, C. Haffey, R. K. Hagmann, S. T. Jackson, A. M. Lynch, E. Q. Margolis, C. Marks, M. D. Meyer, H. Safford, A. D. Syphard,... and J. T. Stevens. 2022. Vegetation type conversion in the US Southwest: Frontline observations and management responses. *Fire Ecology*, 18(1), 6, s42408–022–00131-w. <https://doi.org/10.1186/s42408-022-00131-w>
- Harwell, M. A., J. H. Gentile, L. D. McKinney, J. W. Tunnell, W. C. Dennison, R. H. Kelsey, K. M. Stanzel, G. W. Stunz, K. Withers, and J. Tunnell. 2019. Conceptual Framework for Assessing Ecosystem Health. *Integrated Environmental Assessment and Management*, 15(4), 544–564. <https://doi.org/10.1002/ieam.4152>
- History Colorado. 2024. Ute Treaty of 1868. Ute Treaty of 1868. <https://coloradoencyclopedia.org/article/ute-treaty-1868>
- Houghton, D. C., and R. E. DeWalt. 2023. The caddis aren't alright: Modeling Trichoptera richness in streams of the northcentral United States reveals substantial species losses. *Frontiers in Ecology and Evolution*, 11, 1163922.
- Huffman, D. W., J. E. Crouse, W. W. Chancellor, and P. Z. Fulé,. 2012. Influence of time since fire on pinyon–juniper woodland structure. *Forest Ecology and Management*, 274, 29–37.
- Hummingbird Monitoring Network. 2024. Hummingbird Conservation Networks. Hummingbird Conservation Networks. <https://www.savehummingbirds.org/>
- Jones, H. H., C. Ray, M. Johnson, and R. Siegel. 2024. Breeding birds of high-elevation mixed-conifer forests have declined in national parks of the southwestern US while lower-elevation species have increased, with responses to drought varying by habitat. *Ornithological Applications*, 126(2), 1–16.
- Kaufmann, M. R., D. W. Huisjen, S. Kitchen, M. Babler, S. R. Abella, T. S. Gardiner, D. McAvoy, J. Howie, and D. H. Page, Jr. (2016). Gambel oak ecology and management in the southern Rockies: The status of our knowledge.
- Kellner, K. 2015. jagsUI: A wrapper around 'rjags' to streamline 'JAGS' analyses. CRAN: Contributed Packages. <https://cir.nii.ac.jp/crid/1360864114030096512>
- Kendall, M. G. 1975. Rank correlation methods. <https://psycnet.apa.org/record/1948-15040-000>

- Kéry, M., and M. Schaub. 2011. Bayesian population analysis using WinBUGS: A hierarchical perspective. Academic press.
<https://books.google.com/books?hl=en&lr=&id=Ys6xdkwbywYC&oi=fnd&pg=PP1&dq=K%C3%A9ry,+M.,+and+M.+Schaub.+2012.+Bayesian+population+analysis+using+WinBUGS:+a+hierarchical+perspective.+San+Diego,+USA:+Academic+Press.&ots=5odH8MEh4t&sig=0sfxFu5hHW03NJaZx6gez72An6o>
- Kintsch, J., P. Basting, M. McClure, and J. O. Clarke. 2019. Western slope wildlife prioritization study. <https://trid.trb.org/View/1959650>
- Knight, J., E. Markey, and L. Lopez. 2021. 2021 Vegetation Management Annual Report – Mesa Verde National Park. National Park Service.
- Kochert, M. N., K. Steenhof, C. L. McIntyre, and E. H. Craig. 2002. Golden Eagle (*Aquila chrysaetos*). The Birds of North America Online. <https://doi.org/10.2173/bna.684>
- Koniak, S. 1985. Succession in Pinyon-Juniper Woodlands Following Wildfire in the Great Basin. *The Great Basin Naturalist*, 45(3), 556–566.
- Kruskal, W. H., and W. A. Wallis. 1952. Use of Ranks in One-Criterion Variance Analysis. *Journal of the American Statistical Association*, 47(260), 583–621.
<https://doi.org/10.1080/01621459.1952.10483441>
- Ligon, J. D. 1978. Reproductive Interdependence of Pinon Jays and Pinon Pines. *Ecological Monographs*, 48(2), 111–126. <https://doi.org/10.2307/2937295>
- Ljung, G. M., and G. E. P. Box. 1978. On a measure of lack of fit in time series models. *Biometrika*, 65(2), 297–303. <https://doi.org/10.1093/biomet/65.2.297>
- Mack, R. N. 1986. Alien Plant Invasion into the Intermountain West: A Case History. In H. A. Mooney & J. A. Drake (Eds.), *Ecology of Biological Invasions of North America and Hawaii* (Vol. 58, pp. 191–213). Springer New York. https://doi.org/10.1007/978-1-4612-4988-7_12
- Mann, H. B. 1945. Nonparametric tests against trend. *Econometrica: Journal of the Econometric Society*, 245–259.
- McCauley, R. A., and A. C. Cortés-Palomec. 2023. Taxonomic distinction and population genetic assessment of the Cliff-Palace Milkvetch (*Astragalus deterior* (Barneby) Barneby) in Mesa Verde National Park (Task Agreement P21AC11510 under Cooperative Agreement P18AC00078). Fort Lewis College.
- McClure, E. J., J. D. Coop, C. H. Guiterman, E. Q. Margolis, and S. A. Parks. 2024. Contemporary fires are less frequent but more severe in dry conifer forests of the southwestern United States. *Communications Earth & Environment*, 5(1), 581.

McKinnon, K. A., A. Poppick, and I. R. Simpson. 2021. Hot extremes have become drier in the United States Southwest. *Nature Climate Change*, 11(7), 598–604.

Mesa Verde National Park. 1997. FY 1997 Natural Resources Report (p. 9) [Annual Natural Resources Report].

Mesa Verde National Park. 1998. 1998 Natural Resources Report (p. 18) [Annual Natural Resources Report]. Mesa Verde National Park.

Mesa Verde National Park. 1999. 1999 Natural Resources Report (p. 4) [Annual Natural Resources Report].

Mesa Verde National Park. 2000. FY 2000 Natural Resources Report. Mesa Verde National Park.

Mesa Verde National Park. 2001. FY 2001 Natural Resources Report (p. 39) [Annual Natural Resources Report]. Mesa Verde National Park.

Mesa Verde National Park. 2002. 2002 Natural Resources Report (p. 35) [Annual Natural Resources Report]. Mesa Verde National Park.

Mesa Verde National Park. 2003. 2003 Natural Resources Report (p. 31) [Annual Natural Resources Report]. Mesa Verde National Park.

Mesa Verde National Park. 2004. 2004 Natural Resources Report (p. 29) [Annual Natural Resources Report]. Mesa Verde National Park.

Mesa Verde National Park. 2005. 2005 Natural Resources Report (p. 47) [Annual Natural Resources Report]. Mesa Verde National Park.

Mesa Verde National Park. 2006. 2006 Natural Resources Report (p. 46) [Annual Natural Resources Report]. Mesa Verde National Park.

Mesa Verde National Park. 2007. 2007 Natural Resources Report (p. 53) [Annual Natural Resources Report]. Mesa Verde National Park.

Mesa Verde National Park. 2008. 2008 Natural Resources Report (p. 46) [Annual Natural Resources Report]. Mesa Verde National Park.

Mesa Verde National Park. 2018. Conservation Plan for Chapin Mesa milkvetch (*Astragalus schmollii*) at Mesa Verde National Park. Division of Research and Resource Management, Mesa Verde National Park, Colorado, National Park Service.

Mesa Verde National Park. 2022. Noxious Weed Annual Treatment Plan—2022.

- Mesa Verde National Park (MEVE NP) Colorado. 2024. Climate Change in Mesa Verde. Climate Change.
<https://www.nps.gov/meve/learn/nature/climatechange.htm#:~:text=The%20Pueblo%20people%20experienced%20drought,larger%20and%20more%20frequent%20wildfires>
- Mexican Spotted Owl Recovery Team. 2012. Mexican Spotted Owl Recovery Plan, First Revision (*Strix occidentalis lucida*). Albuquerque, NM: U.S. Fish and Wildlife Service, Southwest Region. Online: <https://www.fs.usda.gov/research/treesearch/45043>
- Miller, R., R. Tausch, and W. Waichler. 1999. Old-growth juniper and pinyon woodlands. SB Mosen and R. Stevens (Compilers). Proceedings: Ecology and Management of Pinyon–Juniper Communities within the Interior West. Proceedings RMRS-P-9, US Department of Agriculture, Forest Service, Rocky Mountain Research Station, Ogden, UT, 375–384.
- Miller, W. J., and D. E. Rees. 2000. Ichthyofaunal Surveys of Tributaries of the San Juan River, New Mexico. Miller Ecological Consultants, Inc.
- Molvar, E. M., R. Rosentreter, D. Mansfield, and G. M. Anderson. 2024. Cheatgrass invasions: History, causes, consequences, and solutions. Ecosphere.
- Moore, L. 2008. Demographic Monitoring of the Cliff Palace Milkvetch (*Astragalus deterior* (Barneby) Barneby): Burned vs. Unburned Populations, Mesa Verde National Park, Southwestern Colorado (p. 37) [Contractor report]. Windom Floristics.
- National Drought Mitigation Center. n.d. Montezuma County, CO | U.S. Drought Monitor. Retrieved July 15, 2024, from https://droughtmonitor.unl.edu/CurrentMap/StateDroughtMonitor.aspx?fips_08083
- National Interagency Fire Center. 2022. InterAgencyFirePerimeterHistory All Years View [Dataset]. <https://data-nifc.opendata.arcgis.com/>
- National Interagency Fire Center. 2024. NFC Open Data Site. NFC Open Data Site. <https://data-nifc.opendata.arcgis.com/>
- National Park Service. n.d. The 24 Associated Tribes of Mesa Verde. <https://www.visitmesaverde.com/media/399909/mesa-verde-associated-tribes.pdf>
- National Park Service. 2015. Foundation Document: Mesa Verde National Park [Foundation Document]. National Park Service.
- National Park Service. 2017. Mesa Verde National Park Trails. <https://public-nps.opendata.arcgis.com/search?collection=Dataset&q=trail>

- National Park Service. 2022. NRSS Key Issues, Stressors, & Threats (KIST) Menu for Park Resource Stewardship Strategies. National Park Service, U.S. Department of the Interior. National Park Service, Dept of Interior.
- National Park Service. 2023. Climate change exposure – Mesa Verde National Park.
- National Park Service. 2024. Fish sampling in Mesa Verde National Park, April 2022. Trip Report (draft). Report to Mesa Verde National Park.
- National Park Service (NPS) Southern Colorado Plateau I&M Network. 2024a. Bird Community Monitoring. Bird Community Monitoring. <https://www.nps.gov/im/scpn/bird-communities.htm>
- National Park Service (NPS) Southern Colorado Plateau I&M Network. 2024b. Upland Vegetation and Soil Monitoring. Upland Vegetation and Soil Monitoring. <https://www.nps.gov/im/scpn/upland-vegetation-and-soil.htm>
- National Park Service (NPS) Southern Colorado Plateau I&M Network. 2024c. Water Quality Monitoring. Water Quality Monitoring. <https://www.nps.gov/im/scpn/water-quality.htm>
- National Weather Service. n.d. Time Series Viewer. NOAA’s National Weather Service. Retrieved October 3, 2024, from <https://www.weather.gov/wrh/timeseries?site=CHAC2>
- Natural Resource Conservation Service. 2023. Ecological site R036XY142CO Loamy Mesa Top (pinyon-Utah juniper). Natural Resource Conservation Service.
- Negrón, J. F., and J. L. Wilson. 2003. Attributes associated with probability of infestation by the piñon ips, *Ips confusus* (Coleoptera: Scolytidae), in piñon pine, *Pinus edulis*. Western North American Naturalist, 440–451.
- Noel, A., D. R. Schlaepfer, B. J. Butterfield, M. C. Swan, J. Norris, K. Hartwig, M. C. Duniway, and J. B. Bradford. 2025. Most Pinyon–Juniper Woodland Species Distributions Are Projected to Shrink Rather Than Shift Under Climate Change. Rangeland Ecology & Management. <https://www.sciencedirect.com/science/article/pii/S1550742424001659>
- Noel, A. R., R. K. Shriver, S. D. Crausbay, and J. B. Bradford. 2023. Where can managers effectively resist climate-driven ecological transformation in pinyon–juniper woodlands of the US Southwest? Global Change Biology, 29(15), 4327–4341. <https://doi.org/10.1111/gcb.16756>
- O’Connor, C. D., D. A. Falk, and G. M. Garfin. 2020. Projected climate-fire interactions drive forest to shrubland transition on an Arizona Sky Island. Frontiers in Environmental Science, 8, 137.
- Olstad, T. 2024a. Condition Summary: Acoustic Environment at Mesa Verde National Park (TAR 33145b). NPS - Natural Sounds & Night Skies Division.

- Olstad, T. 2024b. Condition Summary: Night Skies at Mesa Verde National Park. (TAR 33145a). NPS - Natural Sounds & Night Skies Division.
- Parkyn, S. 2004. Review of Riparian Buffer Zone Effectiveness.
- Pavlacky Jr, D. C., P. M. Lukacs, J. A. Blakesley, R. C. Skorkowsky, D. S. Klute, B. A. Hahn, V. J. Dreitz, T. L. George, and D. J. Hanni. 2017. A statistically rigorous sampling design to integrate avian monitoring and management within Bird Conservation Regions. *PLoS One*, 12(10), e0185924.
- Perry, N. D., P. Morey, and G. San Miguel. 2015. Dominance of a natural water source by feral horses. *The Southwestern Naturalist*, 390–393.
- Phillips, M. L., C. Lauria, T. Spector, J. B. Bradford, C. Gehring, B. B. Osborne, A. Howell, E. E. Grote, R. J. Rondeau, G. M. Trimber, B. Robinson, and S. C. Reed. 2023. Trajectories and tipping points of piñon–juniper woodlands after fire and thinning. *Global Change Biology*, 30(2), e17149. <https://doi.org/10.1111/gcb.17149>
- Pickett, S. T. A., S. T. Pickett, and P. S. White. 1985. *The Ecology of Natural Disturbance and Patch Dynamics*. Academic Press.
- Pinheiro, J., D. Bates, S. DebRoy, S. Sarkar, and R Core Team. 2021. nlme: Linear and Nonlinear Mixed Effects Models. [Computer software]. <https://CRAN.R-project.org/package=nlme>
- Ploughe, L. W., and H. H. Jones. 2024. Southern Colorado Plateau Network Habitat-based bird community monitoring data 2007–2018. [Dataset]. National Park Service. <https://doi.org/10.57830/2302328>
- Plummer, M. 2003. JAGS: A program for analysis of Bayesian graphical models using Gibbs sampling. *Proceedings of the 3rd International Workshop on Distributed Statistical Computing*, 124(125.10), 1–10. <https://www.r-project.org/conferences/DSC-2003/Drafts/Plummer.pdf>
- Pohlert, T. 2024. Non-Parametric Trend Tests and Change-Point Detection. *Non-Parametric Trend Tests and Change-Point Detection*. <https://rdrr.io/cran/trend/f/inst/doc/trend.pdf>
- R Core Team. 2023. R: A language and environment for statistical computing. [Computer software]. R Foundation for Statistical Computing. <https://www.R-project.org/>
- Redmond, M. D., E. S. Golden, N. S. Cobb, and N. N. Barger. 2014. Vegetation Management Across Colorado Plateau BLM Lands: 1950–2003. *Rangeland Ecology and Management*, 67(6), 636–640. <https://doi.org/10.2111/REM-D-13-00171.1>
- Redmond, M. D., A. K. Urza, and P. J. Weisberg. 2023. Managing for ecological resilience of pinyon–juniper ecosystems during an era of woodland contraction. *Ecosphere*, 14(5), e4505. <https://doi.org/10.1002/ecs2.4505>

- Reese, J., M. F. McLaren, J. M. Timmer, M. Smith, T. Walker, C. M. White, Q. Latif, D. C. Pavlacky Jr., and R. A. Sparks. 2023. Integrated Monitoring in Bird Conservation Regions. https://bird-conservancy-of-the-rockies.github.io/IMBCR_AnnualReport_2022/
- Reinhardt, J. R., S. Filippelli, M. Falkowski, B. Allred, J. D. Maestas, J. C. Carlson, and D. E. Naugle. 2020. Quantifying Pinyon-Juniper Reduction within North America's Sagebrush Ecosystem. *Rangeland Ecology and Management*, 73(3), 420–432. <https://doi.org/10.1016/j.rama.2020.01.002>
- Rondeau, R. 2019. Long-term Population Trends and Environmental Attributes of the Imperiled Chapin Mesa Milkvetch (*Astragalus schmolliae*) 2019 Report. Colorado Natural Heritage Program, Colorado State University. https://cnhp.colostate.edu/download/documents/2020/MEVE2019MonitoringReport_final_external.pdf
- Rondeau, R. 2023. Monitoring the Imperiled Chapin Mesa Milkvetch (*Astragalus schmolliae*): 2022 Progress Report on Long-term Population Trends and Environmental Attributes. Colorado Natural Heritage Program, Colorado State University.
- Rondeau, R. J., D. G. Anderson, J. E. Handwerk, and T. Spector. 2022. Post-Fire Effects on the Globally Imperiled Chapin Mesa Milkvetch (*Astragalus schmolliae*), 2001–2019. *Natural Areas Journal*, 42(1), 4–17. <https://doi.org/10.3375/21-14>
- Rondeau, R., K. Decker, J. Handwerk, J. Siemers, L. Grunau, and C. Pague. 2011. The state of Colorado's biodiversity. Prepared for The Nature Conservancy by the Colorado Natural Heritage Program, Colorado State University, Fort Collins, Colorado.
- Royle, J. A. 2004. N-mixture models for estimating population size from spatially replicated counts. *Biometrics*, 60(1), 108–115.
- Sauer, J. R., D. K. Niven, J. E. Hines, D. J. Ziolkowski Jr, K. L. Pardieck, J. E. Fallon, and W. A. Link. 2019. The North American Breeding Bird Survey, Results and Analysis 1966–2019. Version 2.07.2019. <https://www.mbr-pwrc.usgs.gov/>
- Schwinning, S., J. Belnap, D. R. Bowling, and J. R. Ehleringer. 2008. Sensitivity of the Colorado Plateau to change: Climate, ecosystems, and society. *Ecology and Society*, 13(2). <https://www.jstor.org/stable/26268003>
- Sen, P. K. 1968. Estimates of the regression coefficient based on Kendall's tau. *Journal of the American Statistical Association*, 63, 1379–1389.
- Shapiro, S. S., and M. B. Wilk. 1965. An analysis of variance test for normality (complete samples). *Biometrika*, 52(3–4), 591–611. <https://doi.org/10.1093/biomet/52.3-4.591>

- Shinneman, D. J., S. K. McIlroy, S. A. Poessel, R. L. Downing, T. N. Johnson, A. C. Young, and T. E. Katzner. 2023. Ecological effects of pinyon-juniper removal in the Western United States—A synthesis of scientific research, January 2014–March 2021. US Geological Survey. <https://pubs.usgs.gov/publication/ofr20231041/full>
- Sinclair, D. A. 2018. Springs geomorphology influences on physical and vegetation ecosystem characteristics, Grand Canyon Ecoregion, USA [Master's Thesis, Northern Arizona University]. <https://search.proquest.com/openview/37da5bafb6300d1ad258c762e4f52599/1?pq-origsite=gscholar&cbl=18750>
- Smith, D. 2002. Mesa Verde National Park: Shadows of the Centuries (Chapter 6). University Press of Colorado. https://www.nps.gov/parkhistory/online_books/smith/chap6.htm
- Sogge, M. K., R. M. Marshall, S. J. Sferra, and T. J. Tibbitts. 1997. A southwestern willow flycatcher natural history summary and survey protocol. Technical Report NPS/NAUCPRS/NRTR-97/12. USGS Colorado Plateau Research https://www.researchgate.net/profile/Susan-Sferra/publication/252418978_A_Southwestern_Willow_Flycatcher_Natural_History_Summary_and_Survey_Protocol/links/56684cd708ae7dc22ad1db0b/A-Southwestern-Willow-Flycatcher-Natural-History-Summary-and-Survey-Protocol.pdf
- Southern Colorado Plateau Network. 2024. SCPN Water Quality Dataset, 2008–2019 (unpublished). [Dataset]. SCPN.
- Spector, T., K. Puczkowski, T. Clute, A. Baldenweck, and P. Roy. 2019. 2019 Vegetation Management Annual Report – Mesa Verde National Park. National Park Service.
- Stacey, P. B. 2007. Functional Assessment of the Mancos River Watershed.
- Stacey, P. B., A. L. Jones, J. Catlin, D. A. Duff, L. E. Stevens, and C. Gourley. 2021. User's Guide: Rapid Assessment of the Functional Condition of Stream-Riparian Ecosystems in the American Southwest. Sageland Collaborative.
- Stevens, J. T., C. M. Haffey, J. D. Coop, P. J. Fornwalt, L. Yocom, C. D. Allen, A. Bradley, O. T. Burney, D. Carril, M. E. Chambers,... and J. J Walker. 2021. Tamm Review: Postfire landscape management in frequent-fire conifer forests of the southwestern United States. *Forest Ecology and Management*, 502, 119678.
- Stevens, L. E., B. T. Brown, and K. Rowell. 2009. Foraging Ecology of Peregrine Falcons (*Falco peregrinus*) Along the Colorado River, Grand Canyon, Arizona. *The Southwestern Naturalist*, 54(3), 284–299. <https://doi.org/10.1894/MH-19.1>
- Stevens, L. E., J. Jenness, and J. D. Ledbetter. 2020. Springs and Springs-Dependent Taxa of the Colorado River Basin, Southwestern North America: Geography, Ecology and Human Impacts. *Water*, 12(5), Article 5. <https://doi.org/10.3390/w12051501>

- Stowell, S. 2022a. *Astragalus deterior* Demography Monitoring Report 2022 (p. 23) [Monitoring report]. National Park Service.
- Stowell, S. 2022b. Noxious Weed Annual Treatment Plan 2022. National Park Service.
- Stumpf, S. E. 2024. Long- term trend in the Aquatic Macroinvertebrate Community of the Mancos River at Mesa Verde National Park (Monitoring Report 0009-0004-3442–3551; p. 47). National Park Service, Southern Colorado Plateau Network.
- Swan, M. 2024. Vegetation trends in pinyon-juniper woodlands in Southern Colorado Plateau Network parks. NPS - Southern Colorado Plateau Network.
- Thomas, L., M. Hendrie,, C. L. Lauver, S. Monroe, N. J. Tankreto,, S. L. Garman, and M. E. Miller. 2006. Vital signs monitoring plan for the Southern Colorado Plateau Network (Natural Resource Report NPS/SCPN/NRR—2006/002). National Park Service.
- Thomas, K. A., M. L. McTeague, L. Ogden, M. L. Floyd, K. Schulz, B. A. Friesen, T. S. Fancher, R. G. Waltermire, and A. Cully. 2009. Vegetation classification and distribution mapping report: Mesa Verde National Park. In Natural Resource Report (NPS/SCPN/NRR—2009/112). National Park Service. <https://pubs.usgs.gov/publication/70194406>
- Turner, C. E., W. H. Romme, J. Chew, M. E. Miller, L. F-H. Leavesley, G. S. Miguel, N. Cobb, R. Zirbes, R. Vigor, and K. Ironside. 2007. The FRAME project—A collaborative modeling approach to Natural Resource Management at Mesa Verde National Park, Colorado. In *The Colorado Plateau III: Integrating research and resources management for effective conservation*.
- U.S. Department of Agriculture Natural Resource Conservation Service (USDA-NRCS). 1995. Ecological Site Description for Shallow Loamy Mesa Top, Pinyon-juniper Woodland, #141. United States Dept of Agriculture.
- U.S. Fish and Wildlife Service. 2002. Southwestern willow flycatcher recovery plan. U.S. Fish and Wildlife Service.
- U.S. Fish and Wildlife Service. 2022. Endangered and Threatened Wildlife and Plants; Threatened Species Status for Chapin Mesa Milkvetch and Designation of Critical Habitat; Withdrawal. 87(22). <https://doi.org/10.14219/jada.archive.1961.0023>
- U.S. Fish and Wildlife Service. 2023. Mexican Spotted Owl (*Strix occidentalis lucida*) 5-Year Status Review: Summary and Evaluation. [U.S. Fish and Wildlife Service Arizona Ecological Services Office].
- Utah Department of Natural Resources. 2006. Range-wide Conservation Agreement and Strategy for Roundtail Chub (*Gila robusta*), Bluehead Sucker (*Catostomus discobolus*) and Flannelmouth Sucker (*Catostomus latipinnis*). Prepared for the Colorado River Fish and Wildlife Council, by the Division of Wildlife Resources, Salt Lake City, Utah.

- Vermote, E., C. Justice, M. Claverie, and B. Franch. 2016. Preliminary analysis of the performance of the Landsat 8/OLI land surface reflectance product. *Remote Sensing of Environment*, 185, 46–56.
- Weissinger, R., T. E. Philippi, and D. Thoma. 2016. Linking climate to changing discharge at springs in Arches National Park, Utah, USA. *Ecosphere*, 7(10), e01491.
<https://doi.org/10.1002/ecs2.1491>
- Welsh, S., and C. Toft. 1981. Biotic communities of hanging gardens in southeastern Utah. *National Geographic Society Research Reports*, 13, 663–681.
- Wender, B., and K. Doherty. 2011. Long-term post-burn monitoring of former Pinyon-Juniper Woodlands at Mesa Verde National Park. National Park Service.
- Wender, B., and K. Doherty. 2013. Long-term post-burn monitoring of former Pinyon-Juniper Woodlands at Mesa Verde National Park. National Park Service.
- Wethington, S. 2023. Hummingbird Monitoring Network Banding Database (Version I.) [..XI]. Hummingbird Monitoring Network.
- Whitenack, H. 2016. Mapping of Cottonwood Occurrence and Recruitment Post-grazing along the Mancos River in Mesa Verde National Park, Colorado.
- Wickham, H., R. François, L. Henry, K. Müller, and D. Vaughan. 2023. dplyr: A Grammar of Data Manipulation. [Computer software]. <https://github.com/tidyverse/dplyr>
- Wilcox, B. P., L. Turnbull, M. H. Young, C. J. Williams, S. Ravi, M. S. Seyfried, D. R. Bowling, R. L. Scott, M. J. Germino, T. G. Caldwell, and J. Wainwright. 2012. Invasion of shrublands by exotic grasses: Ecohydrological consequences in cold versus warm deserts. *Ecohydrology*, 5(2), 160–173. <https://doi.org/10.1002/eco.247>
- Wilkerson, B. n.d.. Four-letter (English Name) and Six-letter (Scientific Name) Alpha Codes.
- Willey, D. W., and C. VanRiper III. 2000. First year movements by juvenile Mexican spotted owls in the canyonlands of Utah. *Journal of Raptor Research*, 34(1), 1–7.
- Williams, A. P., B. I. Cook, and J. E. Smerdon. 2022. Rapid intensification of the emerging southwestern North American megadrought in 2020–2021. *Nature Climate Change*, 12(3), 232–234.
- Yang, H., X. Yang, M. Heskell, S. Sun, and J. Tang. 2017. Seasonal variations of leaf and canopy properties tracked by ground-based NDVI imagery in a temperate forest. *Scientific Reports*, 7(1), 1267.

- Yelenik, S. G., and J. M. Levine. 2010. Processes Limiting Native Shrub Recovery in Exotic Grasslands after Non-Native Herbivore Removal. *Restoration Ecology*, 18(s2), 418–425.
<https://doi.org/10.1111/j.1526-100X.2009.00598.x>
- Zafar, K. 2019. Cultural Uses of Pinyon and Juniper Forests. Grand Canyon Trust.
<https://www.grandcanyontrust.org/blog/cultural-uses-pinyon-and-juniper-forests>
- Zwank, P. J., S. R. Najera, and M. Cardenas. 1997. Life history and habitat affinities of meadow jumping mice (*Zapus hudsonius*) in the middle Rio Grande valley of New Mexico. *The Southwestern Naturalist*, 318–322.

Appendix A. Condition Rating Tables

Tables 13–20 in Appendix A represent the Condition Rating Tables used in the condition assessments in Chapter 3 of the report.

Table 13. Condition Rating Table for avian species in MEVE.

Condition Rating	Population Trends of All MEVE Species 2008–2022	Occupancy Rates of Birds In MEVE Compared To BCR-16	Passerine Response to Fire	Passerine Response to Time-Since-Fire	Hummingbird Trends
Good	The majority of species tracked by IMBCR have positive pop. trends from 2008 to 2022.	The majority of species have higher occupancy rates in MEVE than in larger region.	Most (>51%) MEVE passerines increased in density in response to fire.	Most (81–100%) MEVE passerines increased in density as time-since-fire increased.	Trends of the 2 most common species are stable or increasing, without large fluctuations over time.
Good/Fair	–	–	Some (0–50%) MEVE passerines increased in density in response to fire.	Many (61–80%) MEVE passerines increased in density as time-since-fire increased.	Trends of both species appear generally stable despite some fluctuations over time.
Fair	There are roughly equal numbers of species with positive pop. trends of 10%/yr or more, versus negative trends of 10%/yr or more, over study period.	Roughly half of species have higher occupancy rates in MEVE and half of species have higher occupancy rates in the larger region.	Most (>51%) MEVE passerines showed no significant change in density in response to fire.	Some (41–60%) MEVE passerines increased in density as time-since-fire increased.	–
Fair/Poor	–	–	Some (0–50%) MEVE passerines decreased in density in response to fire.	Many (61–80%) MEVE passerines decreased in density as time-since-fire increased.	One of the 2 species appears to be declining in MEVE.
Poor	The majority of species tracked by IMBCR have negative pop. trends from 2008 to 2022.	The majority of species have higher occupancy rates in larger region than in MEVE.	Most (>51%) MEVE passerines decreased in density in response to fire.	Most (81–100%) MEVE passerines decreased in density as time-since-fire increased.	The 2 most common hummingbird species appear to be declining in MEVE.

Table 14. Condition Rating Table for Chapin Mesa milkvetch in MEVE. Thresholds for fruit production condition rating classes were developed through consultation with Renee Rondeau, Colorado Natural Heritage Program. Thresholds for seedling density condition rating classes were developed through consultation with Renee Rondeau, CNHP.

Condition Rating	Population Trend (Unburned)	Population Trend (Burned)	Demography (Unburned)	Demography (Burned)	Fruit Production (Unburned)	Fruit Production (Burned)
Good	Chapin Mesa unburned population stable or increasing over time.	Chapin Mesa burned population stable or increasing over time.	All age classes present, and no significant differences between relative abundances of age classes.	All age classes present, and no significant differences between relative abundances of age classes.	>10 fruits/m ²	>10 fruit/m ²
Good/Fair	–	–	All age classes present, but 1–2 incidences of significant differences between relative abundances of age classes.	All age classes present, but 1–2 incidences of significant differences between relative abundances of age classes.	6–10 fruits/m ²	6–10 fruits/m ²
Fair	A slight decrease in trend (observed, but not significant).	A slight decrease in trend (observed, but not significant).	All age classes present, but 2–3 cases of significant differences between relative abundances of age classes.	All age classes present, but 2–3 cases of significant differences between relative abundances of age classes.	4–6 fruits/m ²	4–6 fruits/m ²
Fair/Poor	–	–	All age classes present, but 3–4 cases of significant differences between relative abundances of age classes.	All age classes present, but 3–4 cases of significant differences between relative abundances of age classes.	0–4 fruits/m ²	0–4 fruits/m ²
Poor	Chapin Mesa unburned population showing negative trends over time.	Chapin Mesa burned population showing negative trends over time.	When last assessed in unburned areas, one or more age classes were missing from the population.	When last assessed in burned areas, one or more age classes were missing from the population.	No fruit	No fruit

Table 15. Condition Rating Table for Chapin Mesa milkvetch in MEVE. Thresholds for fruit production condition rating classes were developed through consultation with Renee Rondeau, Colorado Natural Heritage Program. Thresholds for seedling density condition rating classes were developed through consultation with Renee Rondeau, CNHP.

Condition Rating	Seedling Density (Unburned)	Seedling Density (Burned)	Seedling Survival (Unburned)	Seedling Survival (Burned)
Good	At least one year of good seedling density (>0.45 seedlings/m ²) in past 10 years.	At least one year of good seedling density (>0.45 seedlings/m ²) in past 10 years.	>6%	>6%
Good/Fair	–	–	>5–6%	>5–6%
Fair	–	–	>3–5%	>3–5%
Fair/Poor	–	–	>1–3%	>1–3%
Poor	No year of good seedling density (>0.45 seedlings/m ²) in past 10 years.	No year of good seedling density (>0.45 seedlings/m ²) in past 10 years.	0–1%	>0–1%

Table 16. Condition Rating Table for water quality in MEVE.

Condition Rating	Inorganic/Non-Mineral WQ Indicators	Nutrient Indicators	Trace Metal and Mineral Indicators
Good	All indicators met or exceeded state standards from 2009 to 2021.	All indicators met or exceeded state standards from 2009 to 2021.	All indicators met or exceeded state standards from 2009 to 2021.
Good/Fair	>=75% of indicators met or exceeded state standards from 2009 to 2021.	>=75% of indicators met or exceeded state standards from 2009 to 2021.	>=75% of indicators met or exceeded state standards from 2009 to 2021.
Fair	>50–75% of indicators met or exceeded state standards from 2009 to 2021.	>50–75% of indicators met or exceeded state standards from 2009 to 2021.	>50–75% of indicators met or exceeded state standards from 2009 to 2021.
Fair/poor	>25–50% of indicators met or exceeded state standards from 2009 to 2021.	>25–50% of indicators met or exceeded state standards from 2009 to 2021.	>25–50% of indicators met or exceeded state standards from 2009 to 2021.
Poor	<25% of indicators met or exceeded state standards from 2009 to 2021.	<25% of indicators met or exceeded state standards from 2009 to 2021.	<25% of indicators met or exceeded state standards from 2009 to 2021.

Table 17. Condition Rating Table for macroinvertebrates in MEVE.

Condition Rating	Flow Trend, 2008–2019	Trend of Tolerant versus Intolerant Taxa, 2008–2019	Trichoptera Trend, 2008–2019	% Samples That Meet CO Threshold for Attainment of WQ Standard for Biota
Good	Upward trend, or no trend found at either US or DS site (in which case we presume no difference from historical hydrograph through Park).	Downward trend, or no trend in relative abundance of moderately tolerant taxa, and upward or no trend in relative abundance of intolerant taxa.	Upward trends, or no trend in relative abundance and species richness of Trichoptera taxa.	95–100% of samples meet the State threshold.
Good/Fair	–	–	–	>85–95% of samples meet the State threshold.
Fair	Negative trend (non-significant) found at transect level for flow velocity for US and/or DS sites.	Either relative abundance of moderately tolerant taxa increased (non-significant), and/or relative abundance of intolerant taxa decreased (non-significant).	Declines (non-significant) in relative abundance and /or species richness of Trichoptera taxa.	>75–85% of samples meet the State threshold.
Fair/Poor	Significant negative trend found at transect level for flow velocity for either US or DS site.	Either relative abundance of moderately tolerant taxa significantly increased, or relative abundance of intolerant taxa significantly decreased.	Significant declines in either relative abundance or species richness of Trichoptera taxa.	65–75% of samples meet the State threshold.
Poor	Significant negative trend found at transect level for flow velocity for both US and DS sites.	Relative abundance of moderately tolerant taxa significantly increased, and relative abundance of intolerant taxa significantly decreased.	Significant declines in both relative abundance and species richness of Trichoptera taxa.	Less than 65% of samples meet the State threshold.

Table 18. Condition Rating Table for fish and fish/aquatic habitat in Mancos River in MEVE.

Condition Rating	Current Fish Community Diversity (Using CP&W Long-Term Fish Survey Dataset)	Habitat Quality (Using Fish/Aquatic Indicator Score from Most Recent RSRA Survey)	Flow (Using Closest Stream Gage for Longest Period of Record)
Good	Biodiversity reflects pristine or near-pristine conditions with the full suite of native species present and no non-native species.	Score of 4 or above from the most recent RSRA survey.	For most recent period of record, no significant negative trends in either minimum or maximum flow over time are detected, indicating that current hydrograph is likely similar to the historic hydrograph.
Good/Fair	Relative abundances of natives fishes are greater than non-natives, and with a balance of natives that reflect historic relative abundances.	Score between 3 and 4.	–
Fair	Relative abundances of native and non-native fish are similar, or relative abundances of natives are greater than non-natives, but unbalanced in terms of relative abundances of natives.	Score of around 3.	Either minimum or maximum flows over most recent period of record reveal a significant negative trend.
Fair/Poor	Relative abundances of non-native fish are greater than native fish.	Score between 2 and 3.	–
Poor	All fish below 7,000 feet in the Mancos River are non-native. No natives are present.	Score less than 2 from most recent RSRA surveys.	Significant negative trends in both minimum and maximum flow over time are detected, indicating that current hydrograph is not similar to the historic hydrograph.

Table 19. Condition Rating Table for riparian river corridor, using the 2024 Rapid Stream Riparian Assessment (RSRA) functional category scores, on MEVE's stretch of the Mancos.

Condition Rating	Fluvial Geomorphology	Riparian Vegetation	Riparian Structure & Wildlife Habitat
Good	Score of 4 or above from most recent RSRA survey.	Score of 4 or above from most recent RSRA survey.	Score of 4 or above from most recent RSRA survey.
Good/Fair	Score between 3 and 4.	Score between 3 and 4.	Score between 3 and 4.

Table 19 (continued). Condition Rating Table for riparian river corridor, using the 2024 Rapid Stream Riparian Assessment (RSRA) functional category scores, on MEVE's stretch of the Mancos.

Condition Rating	Fluvial Geomorphology	Riparian Vegetation	Riparian Structure & Wildlife Habitat
Fair	Score of about 3.	Score of about 3.	Score of about 3.
Fair/Poor	Score between 2 and 3.	Score between 2 and 3.	Score between 2 and 3.
Poor	Average score less than 2.	Score less than 2.	Score less than 2.

Table 20. Condition Rating Table for PJ woodland condition, using SCPN's indicators for trend assessment, PJ Loamy soil site plot data for 2007 to 2023. Thresholds for cover of shrubs, forbs and perennial grasses, as well as expected species richness (number of species potentially present) derived from Ecological Site Descriptions for Shallow Loamy PJ USDA-NRCS (1995) and Loamy Mesa Top PJ (Natural Resource Conservation Service 2023). Thresholds for total cheatgrass cover derived from Bradley et al. (2018). Thresholds for PJ seedling recruitment derived from Floyd et al. (2015). Thresholds for basal area of PJ derived from Miller et al. (1999) and Floyd et al. (2015).

Condition Rating	Cover of Shrubs ^A	Cover of Perennial Grasses	Forb Cover	Annual Grass Cover ^B	Plot Species Richness (1,500 m Plot Scale)	Piñon Seedling Recruitment ^C	Canopy Cover of PJ ^D
Good	Comprise 20% or more of all foliar cover.	Comprise 15% or more of all foliar cover.	Comprise 12.5% or more of all foliar cover	0% total cover of exotic annual grass	20 or more species found in plots	>1.2 seedlings/m ²	over 40% canopy cover
Good/Fair	Comprise 15–20% of all foliar cover.	Comprise 12.5–15% of all foliar cover.	Comprise 7.5–12.5% of all foliar cover.	0–1% cover.	15–19 species found in plots	1–1.2 seedlings/m ²	31–40% canopy cover
Fair	Comprise 10–15% of all foliar cover.	Comprise 7.5–12.5% of all foliar cover.	Comprise 5–7.5% of all foliar cover.	1–5% cover.	10–14 species found in plots	0.8–1 seedlings/m ²	21–30% canopy cover
Fair/Poor	Comprise 5–10% of all foliar cover.	Comprise 2.5–7.5% of all foliar cover.	Comprise 1–5% of all foliar cover.	5–10% cover.	5–9 species found in plots	0.6–0.8 seedlings/m ²	11–20% canopy cover
Poor	Comprise <5% of all foliar cover.	Comprise <2.5% of all foliar cover.	Comprise <1% of all foliar cover	10 % or more total cover	<5 species found in plots	< 0.6 seedlings/m ²	Less than 10% canopy cover

Appendix B. Extended Methodologies

In Appendix B, we described the methodologies of various analyses from Chapter 3 in further detail.

B.1. Passerine Species Density and Responses to Fire and Vegetation Structure

We modeled bird species density and responses to fire across MEVE in areas ranging from present-day old-growth PJ woodland, to recently burned PJ, as well as older-burn, former-PJ now dominated by oak shrubland or non-native grasses. We quantified breeding bird community composition using point-count data gathered in late May of 2022 and 2023 as part of a long-term monitoring program across national parks and monuments in the Southern Colorado Plateau Inventory & Monitoring Network (SCPN). Survey protocol included 7-minute, unlimited-radius point-counts with distance sampling (Buckland et al. 2001). There are 370 point count stations dispersed throughout MEVE, and 680 surveys were conducted between 2022 and 2023.

We extracted vegetation metrics of interest from aerial, discrete-return LiDAR data. All LiDAR data were downloaded from the Colorado Water Conservation Board website (Colorado Water Conservation Board 2024), and we utilized tiles from Fall 2018 and Fall 2019 scans of southwestern Colorado. Quantum Spatial Inc. collected these data using a Riegl VQ1560i. Remotely sensed vegetation structure information was ground-verified using coarse vegetation classifications determined in the field, which were recorded at each point-count site.

To determine the vegetation structure at point-count sites, we used digital surface models by applying a filtering process to the raw point cloud data. We began by removing points that fell outside a specified vertical range, selecting only those with heights between 0 and 20 meters. This step filters points by their z-coordinate values to exclude ground returns and extremely high points, focusing our dataset on the vegetative layer of interest. Following the filtering, the point cloud data was normalized to height above ground using a Digital Elevation Model (DEM). We created this DEM by interpolating ground points (class 2) using a TIN (Triangulated Irregular Network) approach. The normalized data, representing height above ground, was then used to generate a DSM, which gives the vegetation's surface structure across the study area. Using the filtered and normalized point cloud, we calculated canopy cover and vegetation height (>P95) at the pixel level, producing a 10 m gridded output of these metrics across our study area. We defined canopy cover as the proportion of points above 2 meters relative to the total number of points, representing the area covered by upper vegetation layers. To characterize vegetation height, we calculated the 95th percentile of height values, giving an estimate of the tallest vegetation across the study area. We then extracted mean pixel values for these metrics at each point-count station using an 86-meter buffer, as this was the average detection distance at which we observed birds during surveys.

We also used Google Earth Engine (Gorelick et al. 2017) to generate winter and summer cloud-free mosaics of satellite imagery from the Landsat 8 Operational Land Imager (OLI), utilizing Collection 2 surface reflectance data (Vermote et al. 2016). From these mosaics, we calculated the Normalized Difference Vegetation Index (NDVI) utilizing red and near-infrared bands (bands 4 and 5, respectively). We used the difference between summer and winter NDVI to distinguish between

evergreen PJ woodland and deciduous oak shrubland, representing relative deciduousness across the study site (Yang et al. 2017).

To quantify the fire history of a particular survey location, we obtained publicly available fire boundary polygons from the National Interagency Fire Center data store. For “burned” points, we also calculated “time-since-fire” by subtracting the fire date from 2019, when LiDAR data was published. A 3–4-year gap existed between the point count surveys and the vegetation structure data acquisition via LiDAR. During this period, one fire affected two survey locations. To ensure that our vegetation structure data accurately represented the survey conditions, we excluded these two survey points from our analysis. For points that were located within more than one fire perimeter, we recorded time-since-fire relative to the more recent burn, since we expected that the more recent fire would have had impacts on vegetation structure that are more relevant to present-day bird communities

We employed a Bayesian hierarchical model following Amundson et al. (2014) that evaluates covariates on abundance and detection probability while accounting for zero-inflation and spatial autocorrelation. Several sub-models are produced through this framework; specifically, we developed an N-mixture model of abundance (Royle 2004) and detectability sub-models that incorporate distance sampling and time removal (Farnsworth et al. 2002). Because we were interested in quantifying bird-habitat relationships, flyovers (i.e., birds that were not engaging with the habitat but were simply moving through it) were not included in downstream analyses.

We included a complete set of habitat covariates (canopy cover, vegetation height, deciduousness, burned/unburned, and time-since-fire) in a model fit for each of the 14 species independently using JAGS ver. 4.1.0 (Plummer 2003), called remotely via the R package jagsUI (Kellner 2015). We assessed model convergence using the Gelman-Rubin potential scale reduction parameter, R-hat (Gelman and Rubin 1992), and chains were considered converged when R-hat < 1.2 (Kéry and Schaub 2011). Goodness-of-fit for the availability and perceptibility sub-models were evaluated using Bayesian P-values derived from the posterior distributions (Jones et al. 2024). For each parameter we reported the 90% Bayesian credible interval (BCI), and if the BCI for a given covariate’s coefficient did not include zero, the relationship between the covariate and the associated sub-model was considered supported.

B.2. Chapin Mesa Milkvetch Population Trends

Renee Rondeau of the Colorado Natural Heritage Program (CNHP) provided Chapin mesa milkvetch (CMM) adult and immature density estimates (individuals m⁻²) from 2011 to 2023. The burned populations are considered a separate subpopulation from that within the unburned areas (personal communication, Renee Rondeau, CO Natural Heritage Program, December 2023). For further information regarding survey methods, please reference Rondeau (2019). Rondeau calculated CMM dormancy rates for adults and immatures from 2015 to 2021. CMM dormancy rates allow scientists to avoid assuming CMM adult or immatures missing for one year to be dead. Thus, including these dormancy rates enables calculation of more accurate CMM adult and immature population trends. When reporting dormancy, we did not utilize 2022–2023 data because of the likelihood of a two-year

dormancy for some individuals (dormant plants would not be “captured” in the count without a two-year buffer). For further information regarding dormancy calculations, please reference Rondeau (2023).

We utilized regression models to analyze trends of emerged and dormant CMM adults and immatures of the burned and unburned populations from 2015 to 2021. Because this dataset was a time series, hence the assumption of independence between observations for ordinary least squares (OLS) regression models was violated, we instead used Generalized Least Squares (GLS) regression models. Using R (R Core Team 2023) and the nlme package (Pinheiro et al. 2021) we ran separate GLS regression models for the burned and unburned CMM populations using the `gls()` function with adult/immature density as the response operator and year as the term operator. Results from the GLS regression models with a p-value < 0.05 were deemed statistically significant. We tested the assumptions of the GLS regression models. The residuals were confirmed to be normally distributed via the Shapiro-Wilk test (p-value > 0.05) (Shapiro and Wilk 1965), which we implemented using the `shapiro.test()` function in base R (R Core Team 2023). Additionally, we detected no significant autocorrelation through the Ljung-Box test (p-value > 0.05) (Ljung and Box 1978), utilizing the `box.test()` function in base R (R Core Team 2023).

B.3. Chapin Mesa Milkvetch Demography

Renee Rondeau collected data on CMM fruit density estimates (fruit m⁻²) in burned and unburned transects from 2015 to 2023, with the burned populations being considered a separate subpopulation from that within the unburned areas (personal communication, Renee Rondeau, CO Natural Heritage Program, December 2023). For further information regarding fruit survey methods, please reference Rondeau (2019). To analyze the impacts of various environmental variations on CMM fruit density, we utilized Generalized Least Squares (GLS) regression models in R (R Core Team 2023). We used GLS to quantify the relationship of CMM fruit densities with precipitation and temperature deviations (winter and spring) from 2015 to 2023. Because this dataset was a time series, hence the assumption of independence between observations for ordinary least squares (OLS) regression models was violated, we instead used Generalized Least Squares (GLS) regression models. We combined fruit density data from burned and unburned transects because there was little difference in interannual variability between the two populations so both were likely similarly impacted by environmental variation. We then log-transformed combined fruit density utilizing the `log()` function in base R (R Core Team 2023). Precipitation and temperature deviations (%) were respectively calculated from 1971 to 2000 winter and spring averages (National Weather Service n.d.).

Using the nlme package in R (Pinheiro et al. 2021) we ran multiple GLS regression models using the `gls()` function with log-transformed combined fruit density as the response operator and various environmental variations as the term operator as follows:

1. spring precipitation deviation + spring temperature deviation
2. spring precipitation deviation × spring temperature deviation
3. spring temperature deviation

4. spring precipitation deviation
5. winter precipitation deviation + winter temperature deviation
6. winter precipitation deviation $\times$ winter temperature deviation
7. winter temperature deviation
8. winter precipitation deviation
9. spring precipitation deviation + winter temperature deviation
10. spring precipitation deviation + winter precipitation deviation
11. winter temperature deviation + spring temperature deviation.

Because our dataset contained repeated measures, GLS regression models were deemed inappropriate as the dataset would fail to meet assumptions about temporal autocorrelation. Therefore, we applied a compound symmetry correlation structure using the nlme package (Pinheiro et al. 2021) and corCompSymm() function to all GLS regression models to account for observations taken within the same group. Spring temperature deviations significantly predicted fruit density best of all models [calculated via the AIC() function in base R (R Core Team 2023)]. We tested the assumptions of this GLS regression model. The residuals were shown to be normally distributed via the Shapiro-Wilk test ($p\text{-value} > 0.05$) (Shapiro and Wilk 1965), which we implemented using the shapiro.test() function in base R (R Core Team 2023). Additionally, we detected no significant autocorrelation through the Ljung-Box test ($p\text{-value} > 0.05$) (Ljung and Box 1978), utilizing the box.test() function in base R (R Core Team 2023).

Renee Rondeau also collected data on CMM seedling density estimates (seedlings m^{-2}) in burned and unburned populations from 2015 to 2023. For further information regarding seedling survey methods, please reference Rondeau (2019). To analyze the impacts of various environmental variations on CMM seedling density, we utilized Generalized Least Squares (GLS) regression models in R (R Core Team 2023). We utilized only unburned seedling density in our analyses because of the insufficient sample size of seedlings within burned transects. Precipitation and temperature deviations (%) were respectively calculated from 1971 to 2000 winter and spring averages (National Weather Service n.d.).

Using the nlme package in R (Pinheiro et al. 2021) we ran multiple GLS regression models using the gls() function with log-transformed combined seedling density as the response operator and various environmental variations as the term operator as follows:

1. spring precipitation deviation + spring temperature deviation
2. spring precipitation deviation $\times$ spring temperature deviation
3. spring temperature deviation
4. spring precipitation deviation
5. winter precipitation deviation + winter temperature deviation

6. winter precipitation deviation $\times$ winter temperature deviation
7. winter temperature deviation
8. winter precipitation deviation
9. spring precipitation deviation + winter temperature deviation
10. spring precipitation deviation + winter precipitation deviation
11. winter temperature deviation + spring temperature deviation.

Spring precipitation deviations significantly predicted seedling density best of all models [calculated via the AIC() function in base R (R Core Team 2023)]. We tested the assumptions of this GLS regression model. The residuals were shown to be normally distributed via the Shapiro-Wilk test (p -value > 0.05) (Shapiro and Wilk 1965), which we implemented using the shapiro.test() function in base R (R Core Team 2023). Additionally, we detected no significant autocorrelation through the Ljung-Box test (p -value > 0.05) (Ljung and Box 1978), utilizing the box.test() function in base R (R Core Team 2023).

Renee Rondeau provided data on density estimates (individuals m^{-2}) for each CMM age class (seedlings, yearlings, immatures and adults) in the burned and unburned populations from 2015 to 2021 (Figure 25). For further information regarding survey methods, please reference Rondeau (2019). Rondeau calculated CMM dormancy rates for adults and immatures from 2015 to 2021. CMM dormancy rates allow scientists to avoid assuming CMM adult or immature missing for one year to be dead. Thus, including these dormancy rates enables the calculation of more accurate population trends for CMM adult and immatures. When reporting dormancy, we did not utilize 2022–2023 data because of the possibility of a two-year dormancy for some individuals (dormant plants would not be “captured” in the count without a two-year buffer). Dormancy rates were not calculated for seedlings or yearlings. For further information regarding dormancy calculations, please reference Rondeau (2023).

We assessed if there were significant differences in density estimates between four CMM age classes (adults [emerged and dormant], immatures [emerged and dormant], yearlings, and seedlings) in the burned and unburned populations from 2015 to 2021 using a Kruskal-Wallis test (Kruskal and Wallis 1952). Because the data were non-normal, Kruskal-Wallis tests were used instead of ANOVAs. Using base R (R Core Team 2023) and the pairwise.wilcox.test() function, we ran a Kruskal-Wallis test separately on each population with density as the response vector and age class as the grouping vector. A p.adjust.method of “bonferroni” was applied due to the small sample size of the dataset. Results from the Kruskal-Wallis tests with a p -value < 0.05 were deemed statistically significant.

B.4. Mancos River Water Quality

The Southern Colorado Plateau Network (SCPN) provided data on the water quality of the Mancos River from a site on the east side MEVE downstream of where the river enters the park (Southern Colorado Plateau Network 2024). We calculated the average annual values of all water chemistry parameters [Nitrite ($mg\ L^{-1}$), Ammonia ($mg\ L^{-1}$), E. coli (MPN mL^{-1}), Iron ($ug\ L^{-1}$), Aluminum (ug

L⁻¹), Cadmium (ug L⁻¹), Lead (ug L⁻¹), Manganese (ug L⁻¹), Molybdenum (ug L⁻¹), Uranium (ug L⁻¹), Boron (ug L⁻¹), Arsenic (ug L⁻¹), Copper (ug L⁻¹), Nickel (ug L⁻¹), Selenium (ug L⁻¹), Zinc (ug L⁻¹), Mercury (ug L⁻¹), and Chromium (ug L⁻¹)] and core parameters [non-winter temperature (°C), pH, and dissolved oxygen (%)] from 2009 to 2021 (Figures 26 and 27) in R (R Core Team 2023) using the dplyr package (Wickham et al. 2023) and groupby() and summarise() functions. Some survey values were reported with a less-than symbol, and the rest were not. To address this inconsistency, we averaged all reported values. Therefore, our results present a conservative perspective of the water quality at this site. All averaged annual values were then compared to the state standards for Colorado set by the Water Quality Control Commission (Colorado Department of Public Health and Environment, Water Quality Control Commission n.d.). To meet state standards for core parameters, pH must be within the range of 6.5–9, dissolved oxygen above 60.52% and non-winter temperatures below 28.6 °C. To meet state standards for water chemistry parameters, measurements must be below the following values: Nitrite 0.05 mg L⁻¹, Ammonia 0.10 mg L⁻¹, E. coli 630 MPN mL⁻¹, Iron 100 ug L⁻¹, Aluminum 87 ug L⁻¹, Cadmium 1 ug L⁻¹, Lead 4 ug L⁻¹, Manganese 1650 ug L⁻¹, Molybdenum 150 ug L⁻¹, Uranium 300 ug L⁻¹, Boron 750 ug L⁻¹, Arsenic 7.6 ug L⁻¹, Copper 1.74 ug L⁻¹, Nickel 52 ug L⁻¹, Selenium 4.6 ug L⁻¹, Zinc 65 ug L⁻¹, Mercury 0.01 ug L⁻¹, and Chromium 11 ug L⁻¹.

B.5. Mancos River Fish

We acquired the long-term fish survey dataset from Colorado Parks and Wildlife (CPW) for the Mancos River from 1944 to 2022 (Colorado Parks and Wildlife 2023). This is a comprehensive dataset of all known fish surveys on the Mancos River and includes the numbers of fish per species caught and the relative abundance of species per survey effort (i.e., individuals of a species caught per total individuals caught). We used relative abundances of species for analyses, rather than total catch per species, to ameliorate very large swings in total catch in instances of very low catch rates (i.e., perhaps 10 individuals total per survey) during very low flows. In instances where a fish species was documented at multiple surveys in a single year, multiple relative abundances were recorded for that species within the dataset. We also included a 2022 National Park Service fish survey of the Mancos River within MEVE (National Park Service 2024) that captured similar metrics. We cross-checked that no surveys were missing from this combined dataset using a compendium of fish surveys conducted on San Juan River tributaries from the 1960s to 2000 (Miller and Rees 2000). In order to favor capturing fish typically found in a warm-water fishery, and to control for the possibility of high spatial variability in the fish community composition due to hydrology, water quality, geomorphology and water management, survey sites used in our analyses were stratified by elevation: only Mancos River locations lower than 7000 ft, the approximate elevation of the town of Mancos, were selected for the temporal analyses of fish community composition within MEVE (personal communication, Nic Medley, NPS—Water Resource Division, January 2024).

The fish dataset was parsed into two categories: native and non-native fish guilds. The native fish guild included the following species: bluehead sucker (*Catostomus discobolus*), speckled dace (*Rhinichthys osculus*), fathead minnow (*Pimephales promelas*), flannelmouth sucker (*Catostomus latipinnis*), mottled sculpin (*Cottus bairdii*), and roundtail chub (*Gila robusta*). The non-native fish guild included the following species: black bullhead (*Ameiurus melas*), bluegill sunfish (*Lepomis*

macrochirus), brown trout (*Salmo trutta*), channel catfish (*Ictalurus punctatus*), common carp (*Cyprinus carpio*), green sunfish (*Lepomis cyanellus*), largemouth bass (*Micropterus salmoides*), rainbow trout (*Oncorhynchus mykiss*), red shiner (*Cyprinella lutrensis*), Rio Grande chub (*Gila pandora*), sand shiner (*Notropis stramineus*), and yellow perch (*Perca flavescens*).

We analyzed the relative abundances of the native and non-native fish guilds from 1944 to 2022 using Generalized Least Squares (GLS) regression models, given repeated measures in our dataset which violates the assumption of independence between observations required for an ordinary least squares (OLS) regression model. Using R (R Core Team 2023) and the nlme package (Pinheiro et al. 2021), we ran separate GLS regression models for the native and non-native fish guilds using the gls() function with fish relative abundance as the response operator and year as the term operator. Fish relative abundance values were log-transformed via the log() function in base R (R Core Team 2023). A compound symmetry correlation structure using the nlme package (Pinheiro et al. 2021) and corCompSymm() function was applied to all GLS regression models to account for observations taken with the same group, given the first round of GLS regression models failed autocorrelation assumptions without the application of the correlation structure. Results from the GLS regression models with a p-value < 0.05 were deemed statistically significant. We tested the assumptions of the GLS regression models. The residuals were normally distributed via the Shapiro-Wilk test (p-value > 0.05) (Shapiro and Wilk 1965), which we implemented using the shapiro.test() function in base R (R Core Team 2023). Additionally, we detected no significant autocorrelation through the Ljung-Box test (p-value > 0.05) (Ljung and Box 1978), utilizing the box.test() function in base R (R Core Team 2023).

Experts expressed concern that the species speckled dace skewed the dataset towards a healthier population of native fish than what actually exists. To address this concern, we analyzed the trend of the native guild's relative abundances without speckled dace from 1944 to 2022 utilizing the same GLS regression model and assumptions as described above.

We analyzed the relative abundances of all native and non-native species individually from 1944 to 2022 using GLS regression models. Fish relative abundances were log-transformed using the log() function in base R (R Core Team 2023). The nlme package (Pinheiro et al. 2021) in R was used to run separate GLS regression models for each species using the gls() function with fish relative abundance as the response operator and year as the term operator. Results from the GLS regression models with a p-value < 0.05 were deemed statistically significant. We tested the assumptions of the GLS regression models that produced statistically significant results (bluehead sucker and speckled dace). The residuals were normally distributed via the Shapiro-Wilk test (p-value > 0.05) (Shapiro and Wilk 1965), which we implemented using the shapiro.test() function in base R (R Core Team 2023). Additionally, we detected no significant autocorrelation through the Ljung-Box test (p-value > 0.05) (Ljung and Box 1978), utilizing the box.test() function in base R (R Core Team 2023).

B.6. Mancos River Flow Trends

We analyzed the flow data from the closest downstream gage from MEVE on the Mancos River, the Towaoc Gage (USGS Gage # 09371000), on Ute Mountain Ute lands. Despite the Towaoc Gage

being impacted by runoff events on the Mesa which may impact the river through MEVE, the gage provides some information on the Mancos River via a 103 year (1921–2024) hydrograph. We analyzed trends of minimum flow (cubic feet per second) and maximum flow (cubic feet per second) per year at the Towaoc Gage, which have been available since 1975, via Generalized Least Squares (GLS) regression models. We log-transformed both minimum and maximum flows utilizing the `log()` function in base R (R Core Team 2023). GLS regression models were used because this dataset was a time series and there were violations of the assumptions of an ordinary least squares (OLS) regression model. Using R (R Core Team 2023) and the `nlme` package (Pinheiro et al. 2021), we ran separate GLS regression models for the minimum and maximum datasets using the `gls()` function with flow rates as the response operator and year as the term operator. Results from the GLS regression models with a $p\text{-value} < 0.05$ were deemed statistically significant. We tested the assumptions of this GLS regression model. The models were tested for normalcy with a Q-Q plot, which we implemented using the `qqnorm()` and `qqline()` functions in base R (R Core Team 2023). Additionally, we detected no significant autocorrelation through the Ljung-Box test ($p\text{-value} > 0.05$) (Ljung and Box 1978), utilizing the `box.test()` function in base R (R Core Team 2023).

B.7. Non-native Plant Summarizations

We utilized the Rangeland Analysis Platform (RAP) dataset to extract the average percent cover of annual forb and grasses within MEVE boundaries from 1987 to 2023 (Allred et al. 2021). This dataset is modeled from the long-term Landsat collection imagery record trained on 75,000 field plots of rangeland ecosystems, produced annually at 30 m spatial resolution since 1984. We utilized Generalized Least Squares (GLS) regression models to analyze trends in percent cover of annual forbs in MEVE during two periods. Using R (R Core Team 2023) and the `nlme` package (Pinheiro et al. 2021), GLS regression models were applied to prior/during stand-replacing fire (1987–2004) data and post stand-replacing fire (2005–2023) data separately using the `gls()` function with percent annual forbs and grasses in MEVE as the response operator and year as the term operator. Results from the GLS regression models with a $p\text{-value} < 0.05$ were deemed statistically significant. We tested the assumptions of the GLS regression models. The residuals were normally distributed via the Shapiro-Wilk test ($p\text{-value} > 0.05$) (Shapiro and Wilk 1965), which we implemented using the `shapiro.test()` function in base R (R Core Team 2023). Additionally, we detected no significant autocorrelation through the Ljung-Box test ($p\text{-value} > 0.05$) (Ljung and Box 1978), utilizing the `box.test()` function in base R (R Core Team 2023).

Appendix C. American Ornithological Society Four-letter codes and Associated Common and Scientific Names for all Avian Species Included in NRCA Report.

Table 21 in Appendix C documents all bird species found in MEVE. The table includes information on each species' common name, scientific name and four-letter code. The four-letter codes are used instead of full names in Figures 10 and 12 of section 3.1 of report.

Table 21. AOS 4-letter codes and associated common and scientific bird names. Source: Wilkerson n.d.

Four-Letter Code	Common Name	Scientific Name
AMKE	American kestrel	<i>Falco sparverius</i>
AMRO	American robin	<i>Turdus migratorius</i>
ATFL	Ash-throated flycatcher	<i>Myiarchus cinerascens</i>
BEWR	Bewick's wren	<i>Thryomanes bewickii</i>
BCHU	Black-chinned hummingbird	<i>Archilochus alexandri</i>
BHGR	Black-headed grosbeak	<i>Pheucticus melanocephalus</i>
BTYW	Black-throated gray warbler	<i>Setophaga nigrescens</i>
BGGN	Blue-gray gnatcatcher	<i>Polioptila caerulea</i>
BTHU	Black-tailed hummingbird	<i>Anthracothorax nigricollis</i>
BHCO	Brown-headed cowbird	<i>Molothrus ater</i>
BUSH	Bushtit	<i>Psaltiriparus minimus</i>
CHSP	Chipping sparrow	<i>Spizella passerina</i>
CORA	Common raven	<i>Corvus corax</i>
DUFL	Dusky flycatcher	<i>Empidonax oberholseri</i>
GRFL	Gray flycatcher	<i>Empidonax wrightii</i>
GRVI	Gray vireo	<i>Vireo vicinior</i>
GTTO	Green-tailed towhee	<i>Pipilo chlorurus</i>
HAWO	Hairy woodpecker	<i>Leuconotopicus villosus</i>
HOFI	House finch	<i>Haemorhous mexicanus</i>
HOWR	House wren	<i>Troglodytes aedon</i>
JUTI	Juniper titmouse	<i>Baeolophus ridgwayi</i>
LASP	Lark sparrow	<i>Chondestes grammacus</i>
LEGO	Lesser goldfinch	<i>Spinus psaltria</i>
MOBL	Mountain bluebird	<i>Sialia currucoides</i>
MODO	Mourning dove	<i>Zenaida macroura</i>
NOFL	Northern flicker	<i>Colaptes auratus</i>
PISI	Pine siskin	<i>Spinus pinus</i>
PLVI	Plumbeous vireo	<i>Vireo plumbeus</i>

Table 21 (continued). AOS 4-letter codes and associated common and scientific bird names. Source: Wilkerson n.d.

Four-Letter Code	Common Name	Scientific Name
ROWR	Rock wren	<i>Salpinctes obsoletus</i>
SPTO	Spotted towhee	<i>Pipilo maculatus</i>
TUVU	Turkey vulture	<i>Cathartes aura</i>
VESP	Vesper sparrow	<i>Poocetes gramineus</i>
VGSW	Violet-green swallow	<i>Tachycineta thalassina</i>
VIWA	Virginia's warbler	<i>Leithlypis virginiae</i>
WEBL	Western bluebird	<i>Sialia mexicana</i>
WEME	Western meadowlark	<i>Sturnella neglecta</i>
WEWP	Western wood-pewee	<i>Contopus sordidulus</i>
WBNU	White-breasted nuthatch	<i>Sitta carolinensis</i>
WTSW	White-throated sparrow	<i>Zonotrichia albicollis</i>
WOSJ	Woodhouse's scrub jay	<i>Aphelocoma woodhouseii</i>

Appendix D. Population Trends of 53 Birds in MEVE Tracked by BCR, 2008–2022.

Table 22 of Appendix D documents the population trends of 53 bird species in MEVE via the Bird Conservancy of the Rockies (BCR) from 2008 to 2022 (see Pavlacky, Jr et al. 2017 for methods used to determine trends). The table includes the average trend for each species during that time period as well as various other statistics.

Table 22. Population trends of bird species in MEVE tracked by the Bird Conservancy of the Rockies (BCR) from 2008 to 2022, where a population mean trend) of 1 indicates the population is stable, < 1 indicates the population is declining, and > 1 is an increasing population. CV is the coefficient of variation; LCI 95 and UCI 95 are the lower and upper 95% credible intervals (the true estimate lies within the lower and upper 95% credible intervals with 95% probability); and f is the probability the trend is in the direction of the mean (or the authors' confidence in the direction of the trend, not necessarily the magnitude) (Pavlacky Jr et al. 2017).

Species	Mean Trend	CV	LCI 95	UCI95	Median Trend	f	Percent Change/ Year	No of Detections
Northern Mockingbird	0.678	12.645	0.504	0.859	0.677	1	-32.3	25
Lazuli Bunting	0.72	15.576	0.506	0.937	0.723	0.994	-27.7	15
American Robin	0.762	11.267	0.59	0.925	0.768	0.998	-23.2	13
Cliff Swallow	0.771	20.287	0.478	1.092	0.777	0.934	-22.3	6
Dusky Flycatcher	0.832	8.031	0.703	0.962	0.832	0.99	-16.8	27
Green-tailed Towhee	0.832	6.854	0.712	0.941	0.834	1	-16.6	49
Black-headed Grosbeak	0.834	5.744	0.746	0.927	0.831	1	-16.9	42
White-breasted Nuthatch	0.85	7.685	0.727	0.977	0.846	0.985	-15.4	21
Northern Flicker	0.869	6.429	0.765	0.992	0.869	0.985	-13.1	29
Brown-headed Cowbird	0.875	5.264	0.796	0.978	0.873	0.997	-12.7	37
Mountain Bluebird	0.882	5.236	0.79	0.968	0.885	0.997	-11.5	36
Virginia's Warbler ^A	0.884	4.927	0.797	0.966	0.884	0.994	-11.6	57
Broad-tailed Hummingbird	0.902	5.615	0.801	0.99	0.905	0.979	-9.5	36
Chipping Sparrow	0.921	2.11	0.886	0.96	0.92	1	-8	200

^A Shaded rows indicate the cutoff point where % change in population trend/year is less than 10% per year, where trend shifts from negative to positive, and the cutoff point where % change in population trend/year is less than 10% per year.

Table 22 (continued). Population trends of bird species in MEVE tracked by the Bird Conservancy of the Rockies (BCR) from 2008 to 2022, where a population mean trend) of 1 indicates the population is stable, < 1 indicates the population is declining, and > 1 is an increasing population. CV is the coefficient of variation; LCI 95 and UCI 95 are the lower and upper 95% credible intervals (the true estimate lies within the lower and upper 95% credible intervals with 95% probability); and f is the probability the trend is in the direction of the mean (or the authors' confidence in the direction of the trend, not necessarily the magnitude) (Pavlacky Jr et al. 2017).

Species	Mean Trend	CV	LCI 95	UCI95	Median Trend	f	Percent Change/ Year	No of Detections
Mourning Dove	0.936	2.057	0.898	0.973	0.935	1	-6.5	193
Western Wood-Pewee	0.938	7.214	0.811	1.083	0.939	0.822	-6.1	21
Western Tanager	0.96	5.348	0.861	1.057	0.958	0.792	-4.2	38
Plumbeous Vireo	0.961	2.563	0.916	1.011	0.959	0.938	-4.1	119
Black-throated Gray Warbler	0.962	1.669	0.931	0.993	0.962	0.994	-3.8	330
Cassin's Kingbird	0.966	11.329	0.749	1.197	0.963	0.627	-3.7	10
Juniper Titmouse	0.968	2.764	0.917	1.022	0.97	0.907	-3	104
White-throated Swift	0.969	3.67	0.899	1.034	0.968	0.828	-3.2	73
Gray Flycatcher	0.97	2.134	0.935	1.007	0.97	0.946	-3	158
Blue-gray Gnatcatcher	0.976	1.778	0.945	1.015	0.976	0.898	-2.4	254
Western Bluebird	0.976	4.456	0.888	1.055	0.971	0.705	-2.9	75
Common Raven	0.983	3.009	0.923	1.046	0.982	0.738	-1.8	108
Pinon Jay	0.985	11.603	0.771	1.245	0.979	0.572	-2.1	11
Bewick's Wren	0.986	3.052	0.928	1.041	0.986	0.696	-1.4	109
Canyon Wren	0.989	7.931	0.844	1.147	0.988	0.548	-1.2	12
Ash-throated Flycatcher ^A	0.994	2.527	0.948	1.046	0.994	0.598	-0.6	132
House Wren ^A	1.005	3.551	0.939	1.082	1.004	0.557	0.4	79
Pine Siskin	1.006	13.453	0.738	1.302	1.01	0.524	1	10
Black-chinned Hummingbird	1.01	9.176	0.833	1.215	1.01	0.539	1	14
Spotted Towhee	1.034	1.217	1.009	1.059	1.034	1	3.4	544

^A Shaded rows indicate the cutoff point where % change in population trend/year is less than 10% per year, where trend shifts from negative to positive, and the cutoff point where % change in population trend/year is less than 10% per year.

Table 22 (continued). Population trends of bird species in MEVE tracked by the Bird Conservancy of the Rockies (BCR) from 2008 to 2022, where a population mean trend) of 1 indicates the population is stable, < 1 indicates the population is declining, and > 1 is an increasing population. CV is the coefficient of variation; LCI 95 and UCI 95 are the lower and upper 95% credible intervals (the true estimate lies within the lower and upper 95% credible intervals with 95% probability); and f is the probability the trend is in the direction of the mean (or the authors' confidence in the direction of the trend, not necessarily the magnitude) (Pavlacky Jr et al. 2017).

Species	Mean Trend	CV	LCI 95	UCI95	Median Trend	f	Percent Change/ Year	No of Detections
Vesper Sparrow	1.038	8.387	0.883	1.219	1.032	0.663	3.2	21
House Finch	1.04	4.815	0.95	1.136	1.04	0.771	4	29
Turkey Vulture	1.041	11.077	0.811	1.3	1.029	0.623	2.9	6
Woodhouse's Scrub-Jay	1.042	4.055	0.963	1.129	1.04	0.852	4	57
Hairy Woodpecker	1.044	13.051	0.819	1.3	1.034	0.581	3.4	7
Western Kingbird	1.055	13.509	0.789	1.351	1.052	0.654	5.2	6
Violet-green Swallow	1.08	6.232	0.946	1.222	1.078	0.898	7.8	24
Western Meadowlark	1.087	4.087	0.999	1.176	1.089	0.973	8.9	88
Rock Wren ^A	1.119	5.364	1.016	1.239	1.117	0.985	11.7	44
Red-breasted Nuthatch	1.126	14.23	0.819	1.442	1.121	0.787	12.1	5
Lesser Goldfinch	1.128	10.727	0.916	1.408	1.126	0.873	12.6	13
American Kestrel	1.142	14.238	0.851	1.454	1.142	0.791	14.2	6
Black-billed Magpie	1.157	15.134	0.84	1.503	1.148	0.819	14.8	4
Gray Vireo	1.165	3.796	1.076	1.255	1.166	1	16.6	61
Yellow-rumped Warbler	1.173	27.59	0.484	1.713	1.172	0.699	17.2	5
Wild Turkey	1.183	22.156	0.675	1.699	1.18	0.753	18	5
Lark Sparrow	1.191	7.678	1.037	1.38	1.186	0.997	18.6	23
Common Poorwill	1.193	15.89	0.867	1.644	1.179	0.846	17.9	5
Bushtit	1.222	6.944	1.069	1.399	1.211	1	21.1	23

^A Shaded rows indicate the cutoff point where % change in population trend/year is less than 10% per year, where trend shifts from negative to positive, and the cutoff point where % change in population trend/year is less than 10% per year.

Appendix E. Quantification of Vegetation Cover Change in MEVE with the Rangeland Analysis Platform

Introduction & Methods

To evaluate change in coarsely classified vegetation distribution across MEVE, we used the Rangeland Analysis Platform (Allred et al. 2021). This dataset is modeled from the long-term Landsat collection imagery record trained on 75,000 field plots of rangeland ecosystems, produced annually at 30 m spatial resolution since 1984. For this study, we queried data for MEVE from the Google Earth Engine catalog, which includes years 1986–2023. RAP data consist of fractional cover for six categorical vegetation cover classes: bareground, litter, annual forbs and grasses, perennial forbs and grasses, shrubs, and trees. As these data are presented as continuous values (0–100%) rather than discrete categorical labels indicating a singular cover type as representative of a pixel, RAP data provide an opportunity to evaluate subtle, sub-pixel scale change in vegetation composition over time. For the purpose of demonstrating discrete categorical change over time, Figure 45 displays the singular cover class representing plurality fractional cover value for a given year. However, it is worth noting that in the vast majority of cases this plurality category constituted less than 40% cover, suggesting most pixels were truly mixed in composition rather than dominated by a singular cover class. While RAP data are well-established and transferable beyond the perimeter of MEVE, they do not represent the vegetation of MEVE and vicinity at its most granular level (e.g., species diversity) from which precise natural resource management prescriptions may be considered. Model performance evaluation by RAP developers showed variable prediction accuracy by cover class, with perennial forbs and grasses showing the largest mean absolute error in fractional cover estimates at 10.2% followed by annual forbs and grasses at 7.0%. The producers of this dataset caution that RAP should be used as an accessory to local knowledge of vegetation cover and dynamics. However, this dataset is appropriate for a gap analysis because it reveals general trends of interannual change in cover at moderate resolution, and indication of transitions in successional state, including in response to disturbance events (e.g., fire) and long-term environmental pressures (e.g., climate).

We evaluated interannual change by computing change in fractional cover as difference in pixel values across the period of interest. We quantified trends using Sen's slope, a non-parametric univariate statistical test that is less sensitive to outliers than the slope of a simple linear regression (Sen 1968). Sen's slope is computed using the median value of slopes from each pairing of ordinal observations. We used the “trend” package (Pohlert 2024) in programming software R, with a confidence level of 0.95.

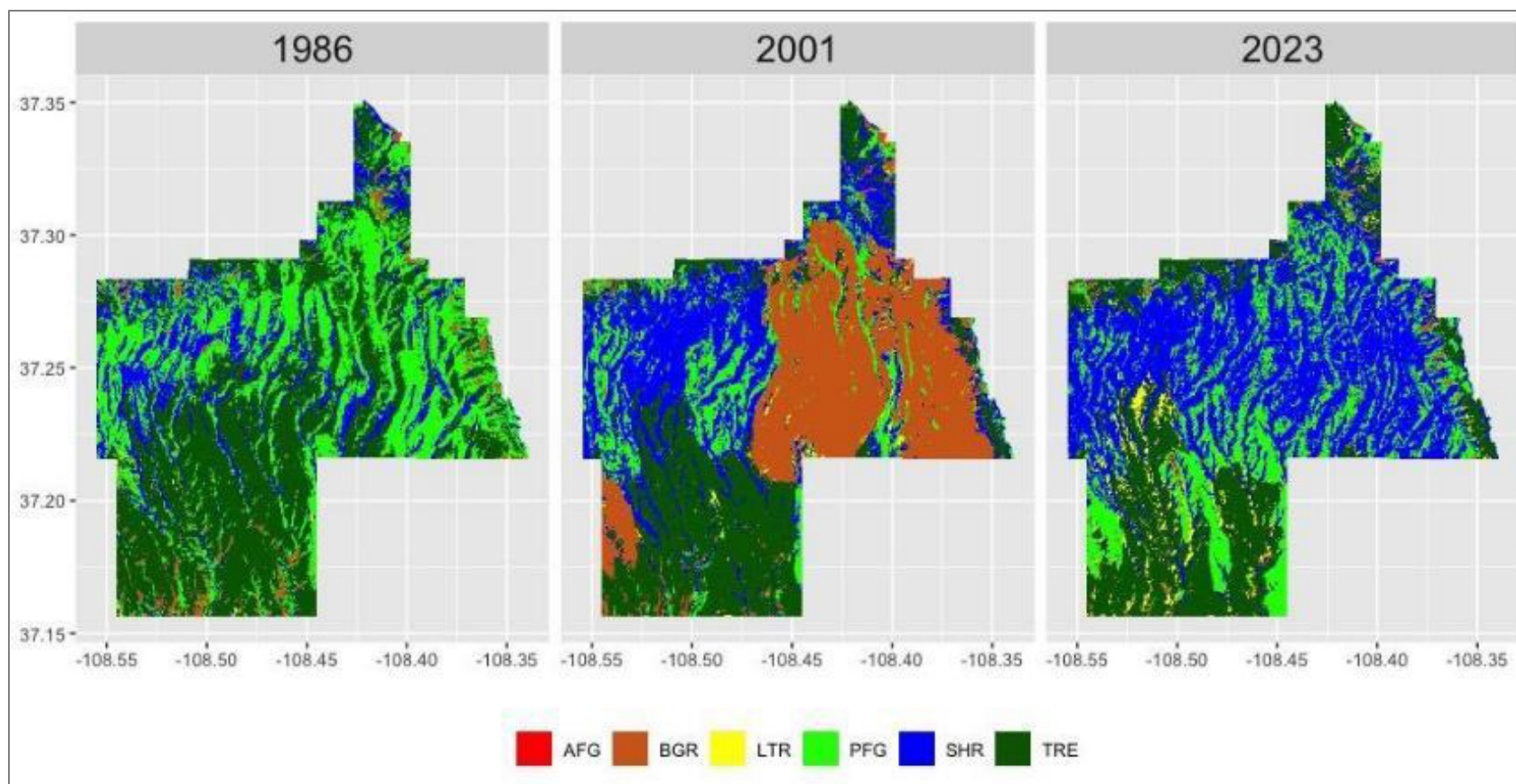


Figure 45. Categorical change in vegetation type (annual forbs and grasses (AFG), bare ground (BGR), litter (LTR), perennial forbs and grasses (PFG), shrub cover (SHR), and tree cover (TRE)) over three years spanning the period of record from which data were processed: 1986, 2001 (post major fires) and 2023. Data were generated from RAP fractional cover, here illustrating the plurality cover class at a 30 m resolution across the MEVE. Created using data from Rangeland Analysis Platform (Allred et al. 2021).

NPS / CONSERVATION SCIENCE PARTNERS

Results

At the Park-scale (i.e., aggregate area), analysis of RAP data revealed a general trend of declining tree cover and increasing shrub cover in MEVE over the period of analysis (Figures 45 and 46). In 1986, the first year of record in our analyses, aggregate tree cover for MEVE was 4,508 ha, peaking at 5,943 ha in 1995 with the lowest observed aggregate cover of 1,980 ha in 2004. Overall, aggregate tree cover declined from 1986 to 3,133 by 2023 (Sen's slope = -73 ha y^{-1}). Aggregate shrub cover showed the greatest variation across the period of record (coefficient of variation (CV) = 0.99), fluctuating between 3,871 ha (1986) and 6,885 ha (2017), and overall trending upward to 6,054 in 2023 (Sen's slope = $+22 \text{ ha y}^{-1}$). Annual forbs and grasses (CV = 0.39) and perennial forbs and grasses (CV = 0.16) showed limited long-term variation and gradually increasing trends (Sen's slope, 1986–2023 = 13 ha y^{-1} and $+12 \text{ ha y}^{-1}$, respectively). Similarly, bare ground (CV = 0.32) and litter (CV = 0.13) showed limited interannual variation and gradual positive trends over the period of analysis (Sen's slope = $+30 \text{ ha y}^{-1}$ and $+25 \text{ ha y}^{-1}$).

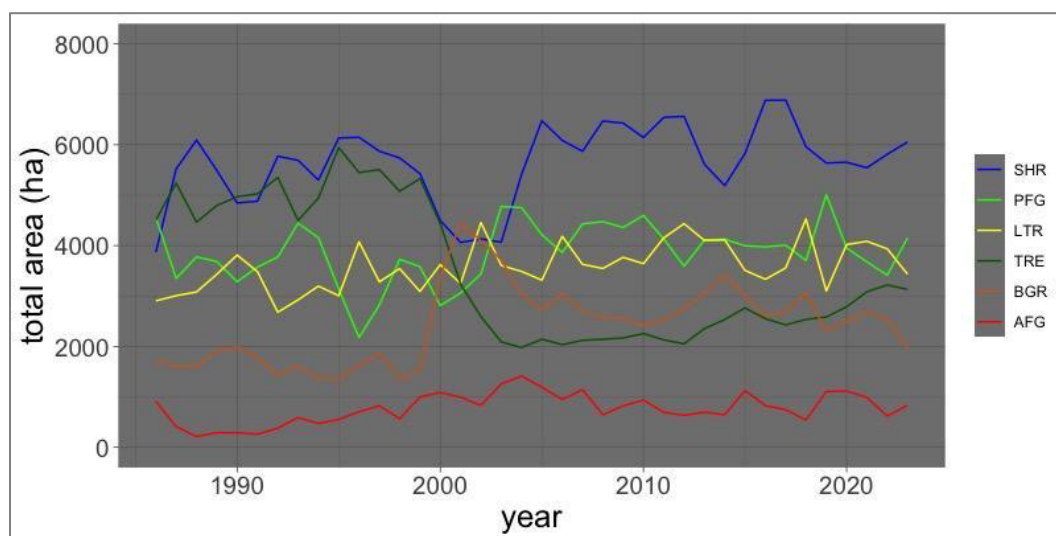


Figure 46. Computed aggregate area by cover class, per year (area of each pixel multiplied by fractional pixel coverage values) for annual forbs and grasses (AFG), bare ground (BGR), litter (LTR), perennial forbs and grasses (PFG), shrub cover (SHR), and tree cover (TRE). Shown as raw values to best illustrate ranked, raw-area coverage as approximated by RAP dataset. Created using data from Rangeland Analysis Platform (Allred et al. 2021).

NPS / CONSERVATION SCIENCE PARTNERS

Below, we present the estimated fractional cover for each of six cover classes for three select years: during the first year of the period of record of RAP (1986), subsequent to the Bircher and Pony fires (2001), and the most recent year available (2023) (Figure 47). Figure 48 conveys the Sen's slope of the trend in fractional cover change of each of the cover classes for three decadal periods (1990–1999, 2000–2009, 2010–2019) within the RAP period of record. We also analyzed change in the cover classes over time, for the five largest fires that have occurred in MEVE between 1989 and 2002. Figures 47–52 depict change in fractional cover over time of annual grasses and forbs, bare ground, litter, perennial grasses and forbs, shrub cover and tree cover within the perimeters of five

major fires in MEVE: Long Mesa (1989); Chapin 5 (1996); Bircher (2000); Pony (2000); and Long Mesa (2002).

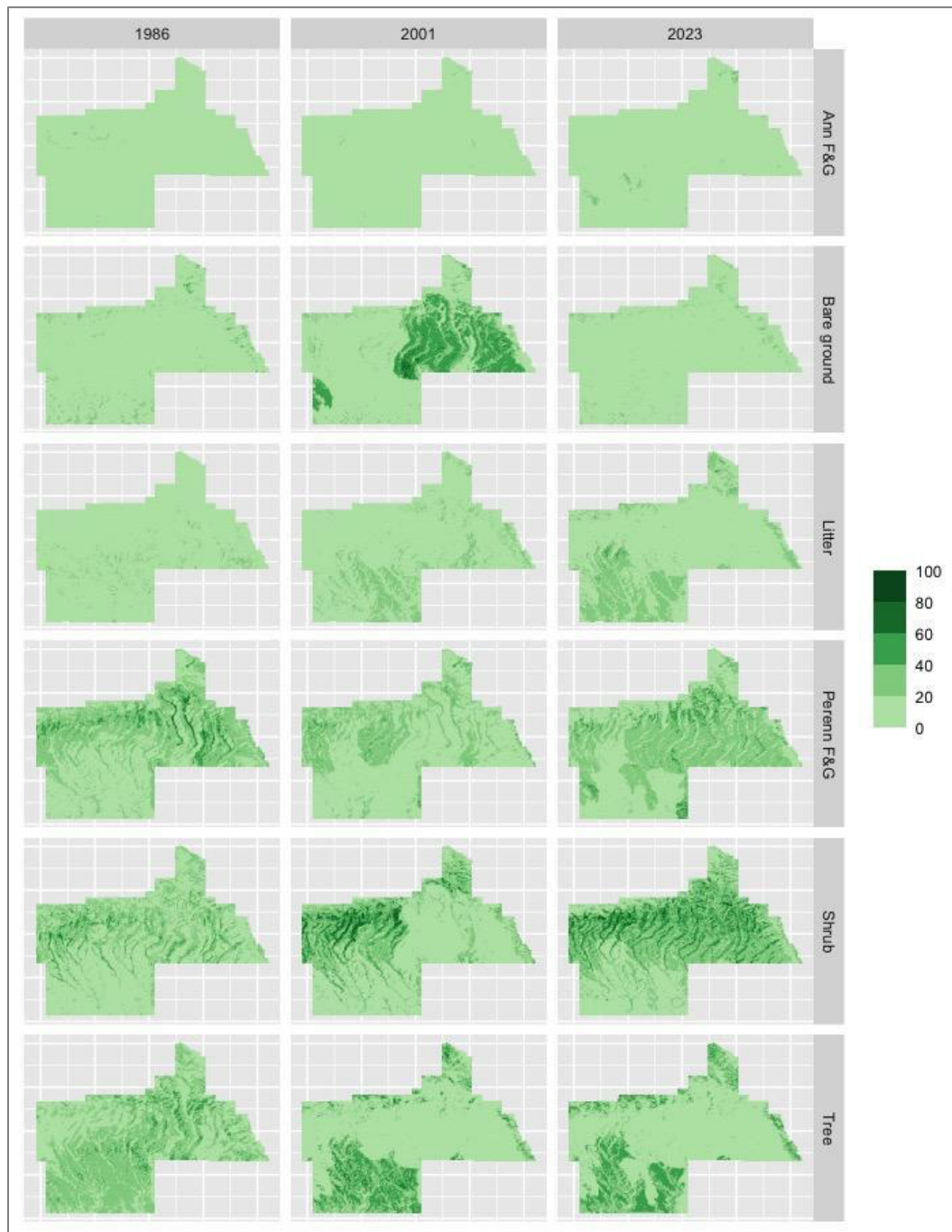


Figure 47. Fractional cover for each of six classes for three select years during the first year of the period of record of RAP (1986), subsequent to the Bircher and Pony fires (2001), and most recent year available (2023). Created using data from Rangeland Analysis Platform (Allred et al. 2021).

NPS / CONSERVATION SCIENCE PARTNERS

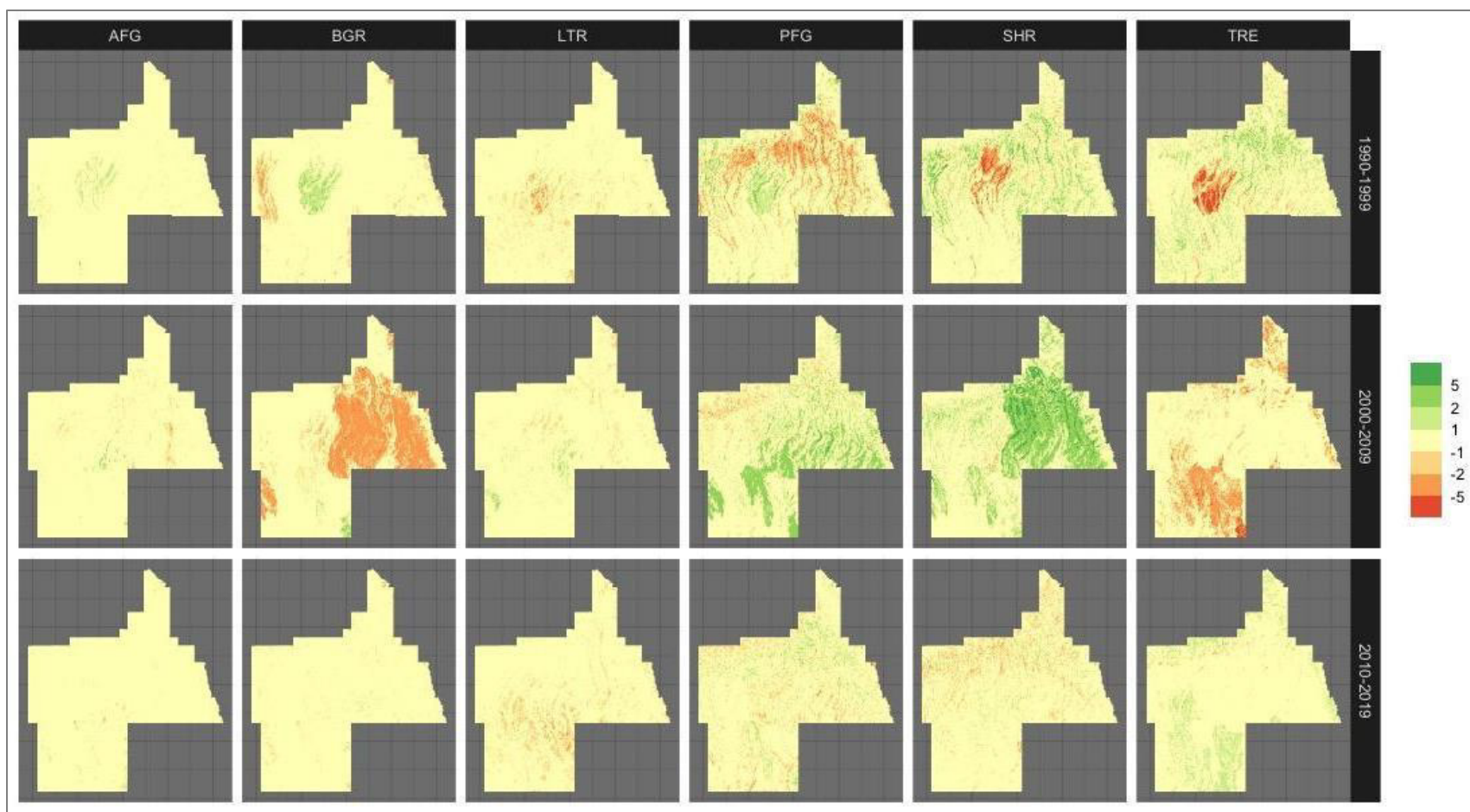


Figure 48. Sen's slope of the trend in fractional cover for each of annual forbs and grasses (AFG), bare ground (BGR), litter (LTR), perennial forbs and grasses (PFG), shrub cover (SHR), and tree cover (TRE) for three decadal periods (1990–1999, 2000–2009, 2010–2019). Created using data from Rangeland Analysis Platform (Allred et al. 2021).

NPS / CONSERVATION SCIENCE PARTNERS

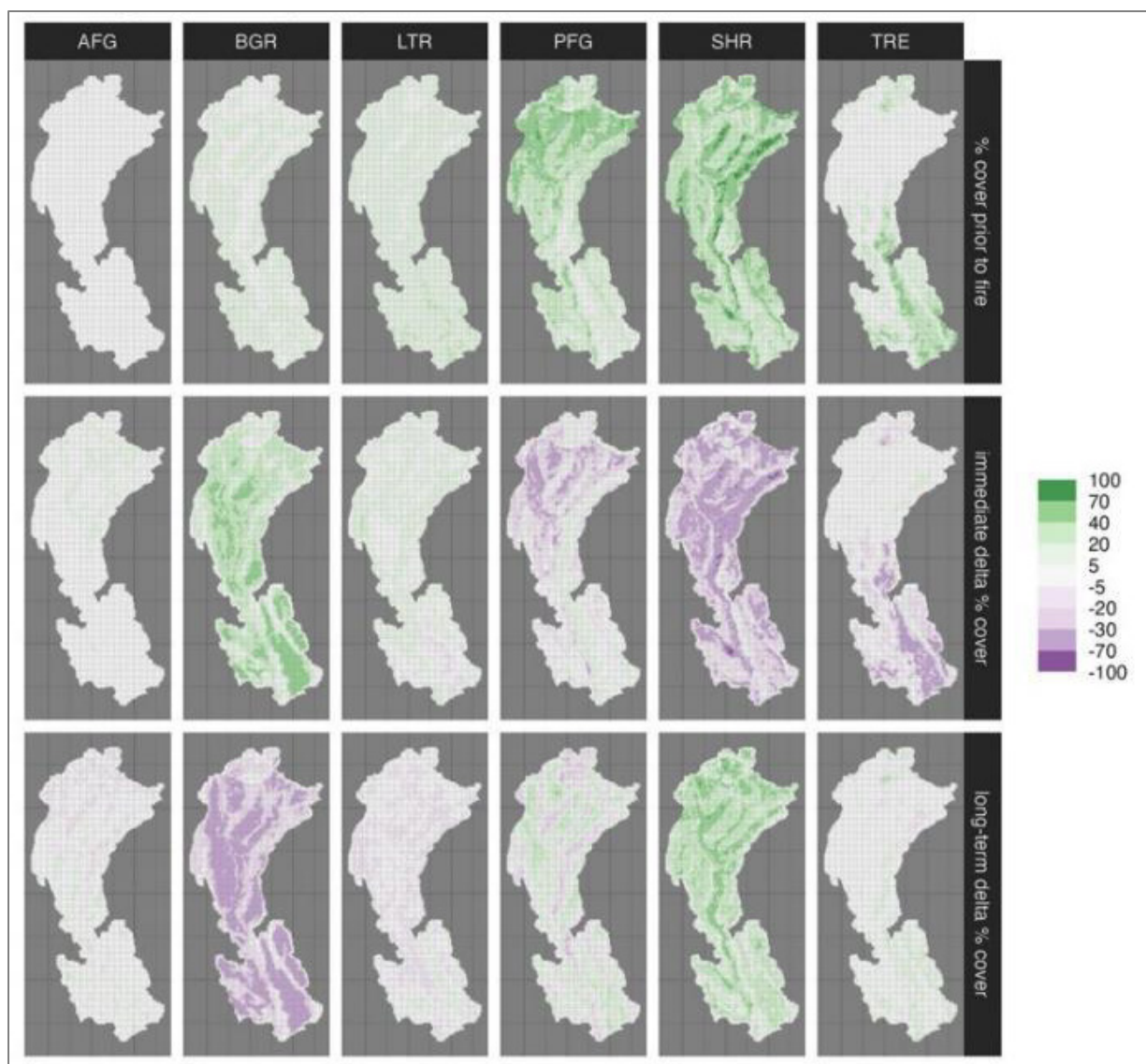


Figure 49. Change in fractional cover of annual forbs and grasses (AFG), bare ground (BGR), litter (LTR), perennial forbs and grasses (PFG), shrub cover (SHR), and tree cover (TRE) within the perimeters of Long Mesa fire (1989). Rows corresponds to observed fractional cover or change in fractional cover relative to pre-fire average for a subset period of years: fractional cover from the year-prior-to fire; change from one-year-prior-to- through one-year-post-fire; and change from one-year-post-fire through the end of period of record (2023). Created using data from Rangeland Analysis Platform (Allred et al. 2021).

NPS / CONSERVATION SCIENCE PARTNERS

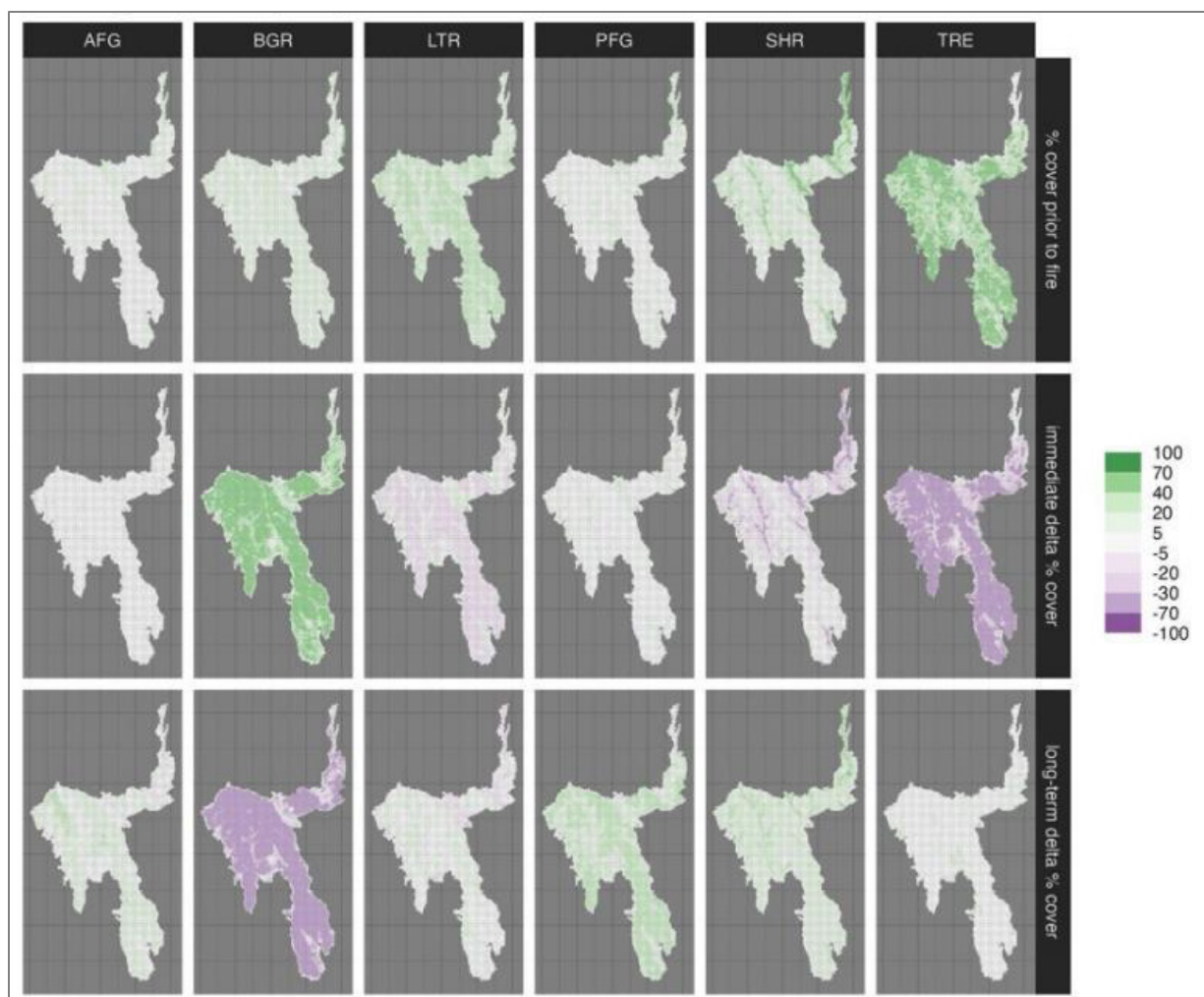


Figure 50. Change in fractional cover of annual forbs and grasses (AFG), bare ground (BGR), litter (LTR), perennial forbs and grasses (PFG), shrub cover (SHR), and tree cover (TRE) within the perimeters of Long Mesa fire (2002). Rows corresponds to observed fractional cover or change in fractional cover relative to pre-fire average for a subset period of years: fractional cover from the year-prior-to fire; change from one-year-prior-to- through one-year-post-fire; and change from one-year-post-fire through the end of period of record (2023). Created using data from Rangeland Analysis Platform (Allred et al. 2021).

NPS / CONSERVATION SCIENCE PARTNERS

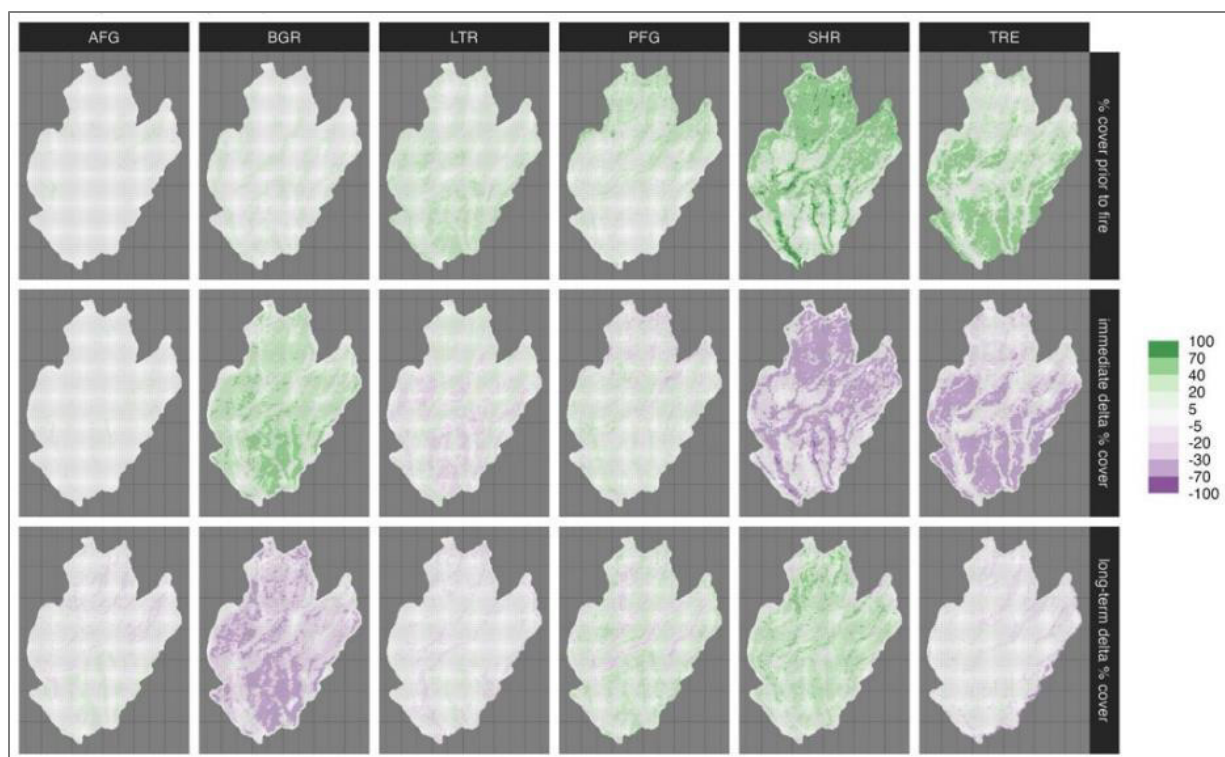


Figure 51. Change in fractional cover of annual forbs and grasses (AFG), bare ground (BGR), litter (LTR), perennial forbs and grasses (PFG), shrub cover (SHR), and tree cover (TRE) within the perimeters of Chapin 5 fire (1996). Rows corresponds to observed fractional cover or change in fractional cover relative to pre-fire average for a subset period of years: fractional cover from the year-prior-to fire; change from one-year-prior-to- through one-year-post-fire; and change from one-year-post-fire through the end of period of record (2023). Created using data from Rangeland Analysis Platform (Allred et al. 2021).

NPS / CONSERVATION SCIENCE PARTNERS

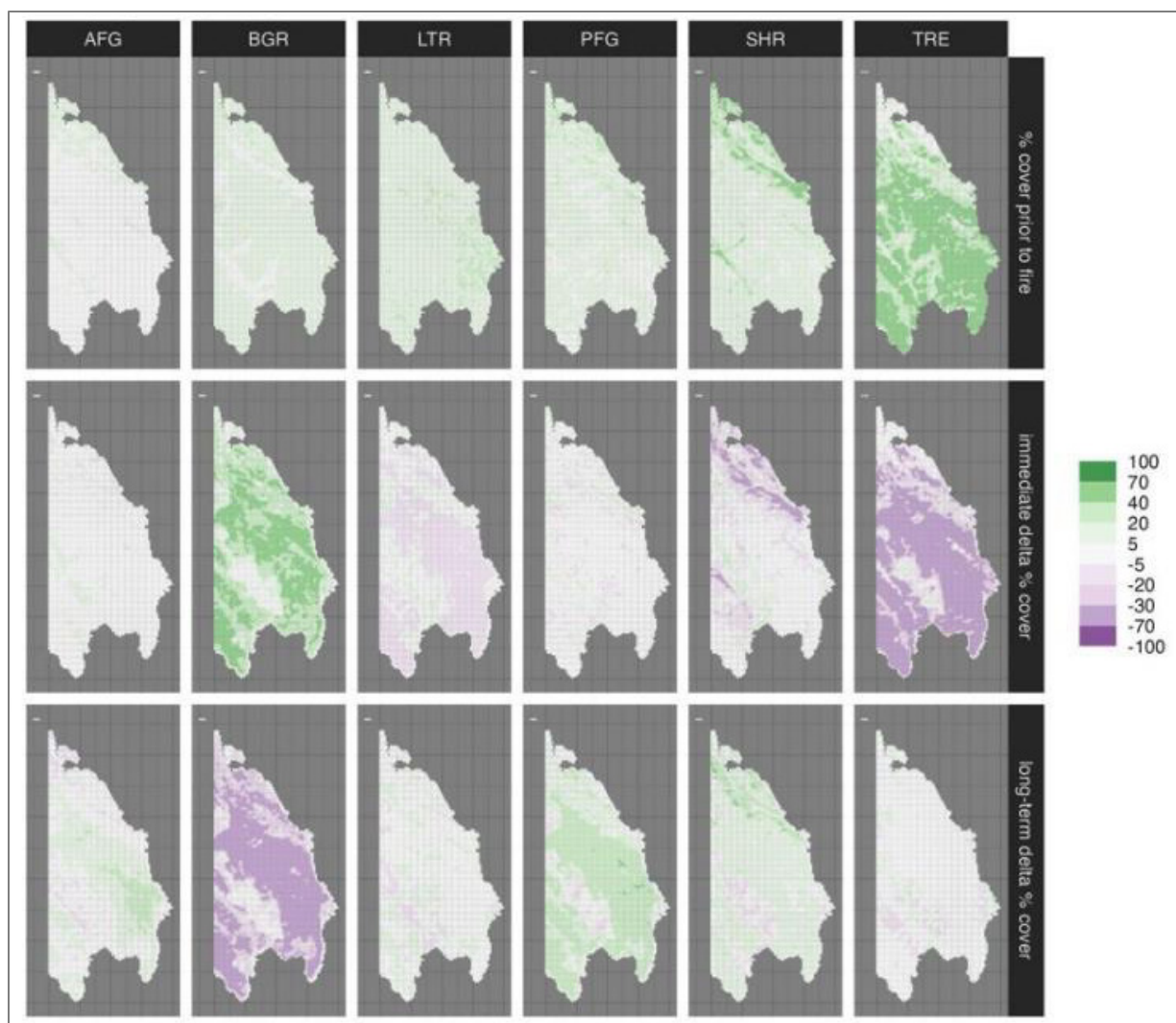


Figure 52. Change in fractional cover of annual forbs and grasses (AFG), bare ground (BGR), litter (LTR), perennial forbs and grasses (PFG), shrub cover (SHR), and tree cover (TRE) within the perimeters of Pony fire (2000). Rows corresponds to observed fractional cover or change in fractional cover relative to pre-fire average for a subset period of years: fractional cover from the year-prior-to fire; change from one-year-prior-to- through one-year-post-fire; and change from one-year-post-fire through the end of period of record (2023). Created using data from Rangeland Analysis Platform (Allred et al. 2021).
NPS / CONSERVATION SCIENCE PARTNERS

National Park Service
U.S. Department of the Interior



Science Report NPS/SR—2025/295
<https://doi.org/10.36967/2311540>

Natural Resource Stewardship and Science

1201 Oakridge Drive, Suite 150
Fort Collins, CO 80525